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Provides guidance for each step in the water quality-based toxics control process from screening to compliance monitoring; Section 1 discusses the advantages and disadvantages of two approaches to toxics control: toxicity testing and chemical specific assessment; Section 2, Water Quality Criteria and Standards, contains a brief discussion of water quality standards and the derivation of ambient criteria to attain and maintain standards; Section 3, Effluent Characterization, describes procedures that provide the data needed to make decisions regarding toxic impact. A two-tiered assessment approach is recommended; Section 4, Human Health Hazard Assessment, describes procedures for assessing potential human health hazards associated with effluents; Section 5, Exposure Assessment, describes procedures for estimating exposure to toxicants and calculating effluent requirements based on ambient criteria and exposure; Section 6, Permit Requirments, describes methods to develop water quality-based NPDES permits; Section 7, Compliance Monitoring, provides recommendations for establishing monitoring requirements in NPDES permits; Section 8, Case Example, illustrates how the recommended procedures from each preceding section are applied in a toxic water quality impact situation; There are five appendices to this document. Several of these are more detailed discussions of specific subjects referenced in the text.

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Technical Support Document for Water Quality-based Toxics Control

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Foreword

The U. S. Environmental Protection Agency (EPA) and the State pollution control agencies have been charged with enforcing the laws regarding pollution of the natural environment. Environmental pollution is considered an urgent and continuing problem and, consequently, the laws grant considerable discretion to the control authorities to define environmental goals and develop the means to attain them. Establishing environmental tolerance levels and incorporating them in a decision making process entails a considerable amount of scientific knowledge and judgment. One area where scientific knowledge is rapidly changing concerns the discharge of toxic pollutants to the Nation's surface waters.

This document provides technical guidance for assessing and regulating the discharge of toxic substances to the waters of the United States. It was issued in support of a recent EPA policy initiative involving the application of biological and chemical assessment techniques to control toxic pollution. The recommendations contained in this document are not mandatory and are intended to be suggestions for approaching problems which tend to be complex and site-specific.

This document is expected to be revised periodically to reflect advances in this rapidly evolving area. Comments from users will be welcomed.

James M Conlon

James M. Conlon, Acting Director
Office of Water Regulations
and Standards

Rebecca W. Hanmer

Rebecca W. Hanmer, Director
Office of Water Enforcement
and Permits

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EFFLUENT CHARACTERIZATION WORKGROUP
Rick Brandes, Workgroup Chairman, U.S. EPA Permits Division
Harold Bergman, University of Wyoming
Richard Kimerle, Monsanto Corporation
Donald Mount, U.S. EPA Environmental Research Laboratory—Duluth
Jim Plafkin, U.S. EPA Monitoring and Data Support Division
Lee Tebo, U.S. EPA Region IV

WASTELOAD ALLOCATION WORKGROUP
Jim Plafkin, Workgroup Chairman, U.S. EPA Monitoring and Data Support Division
Kenneth Dickson, North Texas State University
Dominic DiToro, HydroQual, Inc.
Gordon Loewengart, Allied Chemical Corporation
Alan Maki, Exxon Corporation

EXPOSURE ASSESSMENT WORKGROUP
Mark Morris, Workgroup Chairman, U.S. EPA Monitoring and Data Support Division
Thomas Barnwell, U.S. EPA Environmental Research Laboratory—Athens
Charles Delos, U.S. EPA Monitoring and Data Support Division
Lee Dunbar, Connecticut Department of Environmental Protection
Paul Freedman, Limno-Tech Inc.
Tom Gallager, HydroQual, Inc.
Larry Roesner, Camp, Dresser, and McKee, Inc.
John Pawlow, U.S. EPA Criteria and Standards Division
Elizabeth Southerland, U.S. EPA Monitoring and Data Support Division
Nelson Thomas, U.S. EPA Environmental Research Laboratory—Duluth

HUMAN HEALTH WORKGROUP
Richard Bull, U.S. EPA Environmental Research Laboratory—Cincinnati
Lymon Condie, U.S. EPA Environmental Research Laboratory—Cincinnati
Frank Gostomski, U.S. EPA Criteria and Standards Division
Thomas Purcell, U.S. EPA Criteria and Standards Division

CASE EXAMPLES WORKGROUP
Bruce Newton, Workgroup Chairman, U.S. EPA Permits Division
Thomas Fikslin, U.S. EPA Region II
Marshall Hyatt, U.S. EPA Region IV
Robert McGhee, U.S. EPA Region IV
Walter Redmon, U.S. EPA Region V
Andrew Yasinsac, South Carolina Department of Health

Executive Summary

The Technical Support Document (TSD) for Water Quality-based Toxics Control presents recommendations to regulatory authorities when they are faced with the task of controlling the discharge of toxic pollutants to the nation's waters. It provides guidance for each step in the water quality-based toxics control process from screening to compliance monitoring. It also details the current water quality criteria recommended by the U.S. Environmental Protection Agency (EPA) and how those criteria should be applied. The following is a brief synopsis of the guidance provided in the TSD.

Water Quality Standards and Criteria

Where specific numerical criteria for a chemical or biological parameter (such as toxicity) are absent, compliance with the standards must be based on the general narrative criteria and on protection of the designated use (see National Policy, Appendix A). For many pollutants, EPA-recommended criteria may be used, or data on the toxicological effects of the pollutant may be found in the literature or requested of a discharger.

EPA's recommended criteria for whole-effluent toxicity are as follows. To protect aquatic uses against chronic effects, the ambient toxicity should not exceed 1.0 chronic toxic unit (TU_c) to the most sensitive of at least three test species. For acute effects, the ambient toxicity should not exceed 0.3 acute toxic units (TU_a) to the most sensitive of at least three test species. If only two species are tested and used to evaluate compliance with the criteria, the criteria listed above should be divided by 10. This will account for the additional uncertainty involved with using fewer than the recommended three species.

Aquatic impacts are a function not only of magnitude, but also of duration and frequency with which criteria are exceeded. EPA's recommended criteria for both individual toxicants and whole-effluent toxicity are specified as two numbers: the criteria continuous concentration (CCC), applied as a four-day average concentration; and the criteria maximum concentration (CMC), applied as a one-hour average concentration. The frequency with which criteria are allowed to be exceeded depends on site factors as explained in the text.

Effluent Characterization

A screening analysis is recommended to identify potential problem discharges. Because effluent toxicity and chemistry are often unknown, the screening triggers (or decision criteria) are based mostly on effluent dilution in the receiving water. Extensive data collection is not mandatory in order to establish permit limitations. The reasons for collecting additional data are: to evaluate the degree of a water quality impact; to develop variability information if a dynamic model will be used for a wasteload allocation (WLA); and to test several test species in order to identify the most sensitive, thus avoiding the need to use a species sensitivity factor and identifying a species for compliance monitoring purposes.

If additional assessment is desired, it is recommended that a tiered testing program be followed. At each tier, effluent data and exposure data are combined and compared to triggers using uncertainty factors to account for insufficient data. If no impact is projected, no further assessment or control is required. If impact is projected, regulatory authorities may require additional data to be collected to eliminate the use of uncertainty factors or may institute controls. The uncertainty factors and their associated data requirements are explained in Section 3.

Health Hazard Assessment

Health effects from toxic pollutants are divided into two groups. Pollutants that are carcinogenic or mutagenic are termed genotoxic. These pollutants are presumed to have no safe exposure level (the "no threshold" model), but incremental risk levels can be determined based on the potency of the chemical. All other health effects are termed target organ effects. Pollutants that cause target organ effects are presumed to have safe exposure levels. Exposure criteria or risk estimates are generally developed using animal studies. It is generally impractical to develop such quantitative effects data for complex mixtures such as effluents.

For effluent discharges, it is difficult to develop a site-specific risk assessment because the pollutants discharged are unknown, the degree of human exposure is unknown, and (even if these were known) toxicology information is not available for the vast majority of chemicals. Several short term biological tests have been developed, but these tests are difficult or impossible to translate into potential health hazards, and they cannot quantify the degree of toxicity to humans. Therefore, short term tests are recommended only for screening and priority setting. If screening indicates a potential health hazard, the risk assessment should be based on chemical characterization, available toxicological data and structure activity relationships, and an exposure assessment of the target population.

Exposure and Wasteload Allocation

In order to determine an effluent composition that will protect aquatic organisms, the exposure of the aquatic community to the effluent must be determined. Exposure assessment includes an analysis of how much of the water body is subject to the criteria being exceeded, for how long, and how frequently. The first step is to evaluate the effluent plume dispersion. If mixing is not rapid and complete, the wasteload allocation must be based on a mixing zone analysis (where State standards allow a mixing zone). Means to assess dilution at the edge of a mixing zone are described in Section 5.

If mixing is rapid and complete, there are several models that can be used to assess exposure. The simplest is a steady state model that assumes the toxicant or toxicity does not decay. This model uses a mass balance equation of the following form:

$$\frac{\text{concentration in effluent}}{\text{dilution factor}} = \text{concentration in receiving water}$$

where the inputs are steady state estimates usually designed to reflect some "worst case." Deriving an acceptable effluent concentration to protect water quality simply involves rearranging the equation as follows:

$$\text{criteria concentration} \times \text{dilution factor} = \text{maximum acceptable concentration in effluent}$$

Steady state models work on the assumption that the effluent concentration is steady and that the duration and frequency with which criteria are exceeded can be reflected entirely by selecting a critical flow condition in the receiving water of appropriate duration and frequency (see Appendix D).

However, the assumption that effluent concentration is steady state does not hold for most effluents. A second problem is that the acceptable effluent concentration derived from a steady state model does not include any information about acceptable effluent variability. Therefore, an interim toxicologically based steady state modeling approach is recommended to provide some variability information along with the acceptable effluent concentration projections.

The best means of modeling exposure is to use computer models that incorporate the variability of the individual inputs (such as effluent flow and concentration, receiving water flow, temperature, background concentration, etc.). These models are termed dynamic models and are much more accurate than steady state models in reflecting or predicting exposure provided adequate data exist. The acceptable effluent condition derived using these models is expressed as the effluent long term average (LTA) and variance, which greatly simplifies derivation of permit limits. Three dynamic modeling approaches are described along with instructions for their use.

Permit Requirements

The requirements of a WLA must be incorporated in the wastewater discharge permit. In many cases permit limits will be different than the WLA to reflect different assumptions and means of expressing effluent quality. Three types of WLAs are identified, and recommendations are provided for deriving permit limits to properly enforce each type of WLA. Other permit-related issues such as data collection, limit expression, and compliance schedules are discussed.

Compliance Monitoring

For important parameters, at least once per week monitoring is recommended. A phased monitoring approach is discussed in which monitoring frequency is reduced after continuing compliance is demonstrated. Unique considerations for enforcement of water quality-based permits are discussed.

Case Example

A multiple source, "integrated" approach case example based on data from several toxic discharge sites is provided. The decisions made regarding screening, effluent characterization, exposure assessment, and permit limit derivation are described.

List of Abbreviations

μg	microgram
μm	micrometers
AA	atomic absorption
ACR	acute-chronic ratio
ADI	acceptable daily intake
BAT	best available technology
BCF	bioconcentration factor
BCT	best conventional technology
BOD	biological oxygen demand
BPJ	best professional judgment
BPT	best practicable technology
CAG	Carcinogen Assessment Group (EPA)
CCC	criterion continuous concentration
Cd	cadmium
CETIS	Complex Effluent Toxicity Information System
cfs	cubic feet per second
CHNTRN	Channel Transport Model
CIS	Chemical Information System
CMC	criterion maximum concentration
CN	cyanide
Cr^{+6}	chromium
CTAP	Chemical Transport and Analysis Program
Cu	copper
CV	coefficient of variation
DMR	discharge monitoring report
DO	dissolved oxygen
DOC	dissolved organic carbon
EPA	U.S. Environmental Protection Agency
ERL	Environmental Research Laboratory (EPA)
EXAMS	Exposure Analysis Modeling System
FDA	Food and Drug Administration
FETRA	Finite Element Transport Model
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
g	grams
GC	gas chromatography
HPLC	high-pressure liquid chromatography
HSPF	Hydrologic Simulation Program - FORTRAN
IHS	in-house software
IWC	instream waste concentration
kg	kilograms
l	liter
LC50	lethal concentration killing 50% of exposed organisms
LTA	long term average
MEXAMS	Metals Exposure Analysis Modeling System
mg	milligrams
MGD	million gallons per day
MICH	Michigan River Model
ml	milliliter
MS	mass spectroscopy
MSDS	Monitoring and Data Support Division (EPA)
Ni	nickel
NLM	National Library of Medicine
NOEL	no observable effect level
NPDES	National Pollutant Discharge Elimination System
OECD	Organization of Economic Cooperation Development
ORD	Office of Research and Development (EPA)
POTW	publicly owned treatment works
SAR	structure-activity relationships
SERATRA	Sediment Contaminant Transport Model
SLSA	Simplified Lake/Stream Analysis
STP	sewage treatment plant
TMDL	total maximum daily load
TOC	total organic carbon
TODAM	Transport One-Dimensional Degradation and Migration Model
TOXIWASP	Chemical Transport and Fate Model
TOXIC	Toxic Organic Transport and Bioaccumulation Model
TSCA	Toxic Substances Control Act
TSS	total suspended solids
TU_a	acute toxic unit
TU_c	chronic toxic unit
USGS	U.S. Geological Survey
WASTOX	Estuary and Stream Quality Model
WHO	World Health Organization
WLA	wasteload allocation
Zn	zinc

Glossary

acute involving a stimulus severe enough to rapidly induce a response; in toxicity tests, a response observed in 96 hours or less typically is considered acute. An acute effect is not always measured in terms of lethality; it can measure a variety of effects. Note that acute means *short*, not mortality.

acute-chronic ratio (ACR) the ratio of the acute toxicity (expressed as an LC50) of an effluent or a toxicant to its chronic toxicity (expressed as an NOEL). It is used as a factor for estimating chronic toxicity on the basis of acute toxicity data.

additivity the characteristic property of a mixture of toxicants that exhibits a cumulative toxic effect equal to the arithmetic sum of the effects of the individual toxicants.

ambient toxicity toxicity manifested by a sample collected from an aquatic receiving system.

antagonism the characteristic property of a mixture of toxicants that exhibits a less-than-additive cumulative toxic effect.

bioaccumulation uptake and retention of substances by an organism from its surrounding medium and from food.

bioassay a test used to evaluate the relative potency of a chemical by comparing its effect on a living organism with the effect of a standard preparation on the same type of organism. Bioassays are frequently used in the pharmaceutical industry to evaluate the potency of vitamins and drugs. "Bioassay" and "toxicity test" are not synonymous.

bioconcentration uptake of substances from the surrounding medium through gill membranes or other external body surfaces.

bioavailability the property of a toxicant that governs its effect on exposed organisms. A reduced bioavailability would have a reduced toxic effect.

chronic involving a stimulus that lingers or continues for a relatively long period of time, often one-tenth of the life span or more. Chronic should be considered a relative term depending on the life span of an organism. A chronic effect can be lethality, growth, reduced reproduction, etc. Chronic means *long*.

conservative pollutant a pollutant that is persistent and not subject to decay or transformation.

continuous stimulation model a fate and transport model that uses timeseries input data to predict receiving water quality concentrations in the same chronological order as that of the input variables.

criteria continuous concentration (CCC) the EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed indefinitely without causing unacceptable effect.

criteria maximum concentration (CMC) the EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed for a brief period of time without causing mortality.

design flow the critical flow used for steady state wasteload allocation modeling.

critical life stage the period of time in an organism's lifespan in which it is the most susceptible to adverse effect caused by exposure to toxicants, usually during early development (egg, embryo, larvae). Chronic toxicity tests are often run on critical life stages to replace long duration, life cycle tests since the toxic effect occurs during the critical life stage.

diversity the number and abundance of species in a specified location.

duration the period of time over which the instream concentration is averaged for comparison with criteria concentrations. This specification limits the duration of concentrations above the criteria.

effluent biomonitoring the measurement of the biological effects of effluents (such as toxicity, biostimulation, and bioaccumulation).

Final Acute Value (FAV) an acute toxicity limit in which the measured acute toxicity expressed as an LC50 of an effluent or a toxicant is adjusted by a factor (0.3 is recommended) to eliminate mortality.

frequency how often criteria can be exceeded without unacceptably affecting the community.

LC50 the toxicant concentration killing 50% of exposed organisms at a specific time of observation.

lognormal probabilistic dilution model a dilution model that calculates the probability distribution of receiving water quality concentrations from the lognormal probability distributions of the input variables.

magnitude how much of a pollutant (or pollutant parameter such as toxicity), expressed as a concentration or toxic unit is allowable.

Monte Carlo simulation a stochastic modeling technique that involves the random selection of sets of input data for use in repetitive model runs in order to predict the probability distributions of receiving water quality concentrations.

No Observed Effect Level (NOEL) the highest measured continuous concentration of an effluent or a toxicant that causes no observed effect on a test organism.

permit averaging period the duration of time over which a permit limit is calculated — day(s), week, or month.

persistence that property of a toxicant or an effluent that is a measurement of the duration of its effect. A persistent toxicant or toxicity maintains effect after mixing, degrading slowly. A non-persistent toxicant or toxicity may have a quickly reduced effect after mixing as degradation processes such as volatilization, photolysis, etc., transform the chemical.

plug flow sampling a monitoring procedure that follows the same slug of wastewater throughout its transport in the receiving water. Water quality samples are collected at receiving water stations, tributary inflows, and point source discharges only when a dye slug or tracer passes that point.

probability a number expressing the likelihood of occurrence of a specific event, such as the ratio of the number of outcomes that will produce a given event to the total number of possible outcomes.

probability distribution a mathematical representation of the probabilities that a given variable will have various values.

recurrence interval the average number of years within which a variable will be less than or equal to a specified value. This term is synonymous with *return period.*

steady state model a fate and transport model that uses constant values of input variables to predict constant values of receiving water quality concentrations.

STORET EPA's computerized water quality data base that includes physical, chemical, and biological data measured in waterbodies throughout the United States.

sublethal involving stimulus below the level that causes death.

synergism the characteristic property of a mixture of toxicants that exhibits a greater-than-additive cumulative toxic effect.

total maximum daily load (TMDL) the total allowable pollutant load to a receiving water such that any additional loading will produce a violation of water quality standards.

toxicity test the means to determine the toxicity of a chemical or an effluent using living organisms. A toxicity test measures the degree of response of an exposed test organism to a specific chemical or effluent.

toxic unit acute (TU_a) the reciprocal of the effluent dilution that causes the acute effect by the end of the acute exposure period.

toxic unit chronic (TU_c) the reciprocal of the effluent dilution that causes no unacceptable effect on the test organisms by the end of the chronic exposure period.

uncertainty factors factors used in the adjustment of toxicity data to account for unknown variations. Where toxicity is measured on only one test species, other species may exhibit more sensitivity to that effluent. An uncertainty factor would adjust measured toxicity upward and downward to cover the sensitivity range of other, potentially more or less sensitive species.

wasteload allocation (WLA) the portion of a receiving water's total maximum daily pollutant load that is allocated to one of its existing or future point sources of pollution.

whole-effluent toxicity the aggregate toxic effect of an effluent measured directly with a toxicity test.

Introduction

Purpose

The purpose of this Technical Support Document (TSD) for Water Quality-Based Toxics Control is to provide procedural recommendations for identifying, analyzing, and controlling adverse water quality impacts caused by the discharge of toxic pollutants to the waters of the United States. The guidance supplied in this document is meant to support implementation of the U.S. Environmental Protection Agency's (EPA) national "Policy for the Development of Water Quality-Based Permit Limitations for Toxic Pollutants" (Appendix A).

The recommendations presented in this document are not binding. They have been developed as the most practical recommendations for the current state of knowledge. The field of assessment of water quality impacts from toxic pollutants is evolving and, therefore, recommended techniques can be expected to evolve also. EPA is currently developing detailed guidance related to several aspects of the control of toxic pollutants. These efforts are referenced at appropriate points in the document.

The need for assessment and control of toxic pollutants should not be underestimated. The EPA national policy statement was issued because of evidence that many industrial and municipal sources continue to discharge toxic substances in amounts significant to the waterbody. Data collected through the EPA's ongoing Complex Effluent Toxicity Testing Program indicate that increased attention should be directed toward municipal point sources treating industrial wastewaters.

Development of techniques to assess and control toxics has focused primarily on individual toxicants. Analytical methods and fate and transport models are established to assess the effects of the discharge of specific chemicals. Toxicity, as measured through toxicity testing of effluents, has not been widely used as a control parameter in the National Pollutant Discharge Elimination System (NPDES) permits program. Now, EPA's national policy recommends an integrated approach to controlling toxic pollutants using either or both of these approaches, as appropriate.

Organization of the Technical Support Document

Section 1 discusses the advantages and disadvantages of the two approaches and also includes a discussion of certain issues associated with the assessment and control of toxic pollutants.

The relationship of subsequent sections of the document to the process of water quality management is schematically presented in Figure 1. Each section is noted according to the aspect of the process it covers.

Section 2, Water Quality Criteria and Standards, contains a brief discussion of water quality standards and the derivation of ambient criteria to attain and maintain standards.

Section 3, Effluent Characterization, describes procedures that provide the data needed to make decisions regarding toxic impact. A two-tiered assessment approach is recommended. In Tier 1 (Screening), a process for cost-effective screening is described. In Tier 2 (Definitive Data Generation), a data collection process is described in which increasingly more complete data are generated in stages using uncertainty factors to judge whether or not an effluent exceeds a dilution-based margin of safety.

Section 4, Human Health Hazard Assessment, describes procedures for assessing potential human health hazards associated with effluents. Some testing procedures for measuring bioaccumulation are also presented.

Section 5, Exposure Assessment, describes procedures for estimating exposure to toxicants and calculating effluent requirements based on the ambient criteria and exposure. Exposure is expressed in terms of concentration, area, duration (how long will an exposure level exist), and frequency (how often will an exposure level be experienced).

The analytical techniques are presented in order of complexity, from simple dilution evaluations to more advanced modeling techniques and probability analyses. A discussion of mixing zones is included.

Section 6, Permit Requirements, describes methods to develop water quality-based NPDES permits. Available permitting options for toxics control are described including procedures for the establishment of Toxicity Reduction Evaluations by individual municipal and industrial dischargers.

Section 7, Compliance Monitoring, provides recommendations for establishing monitoring requirements in NPDES permits.

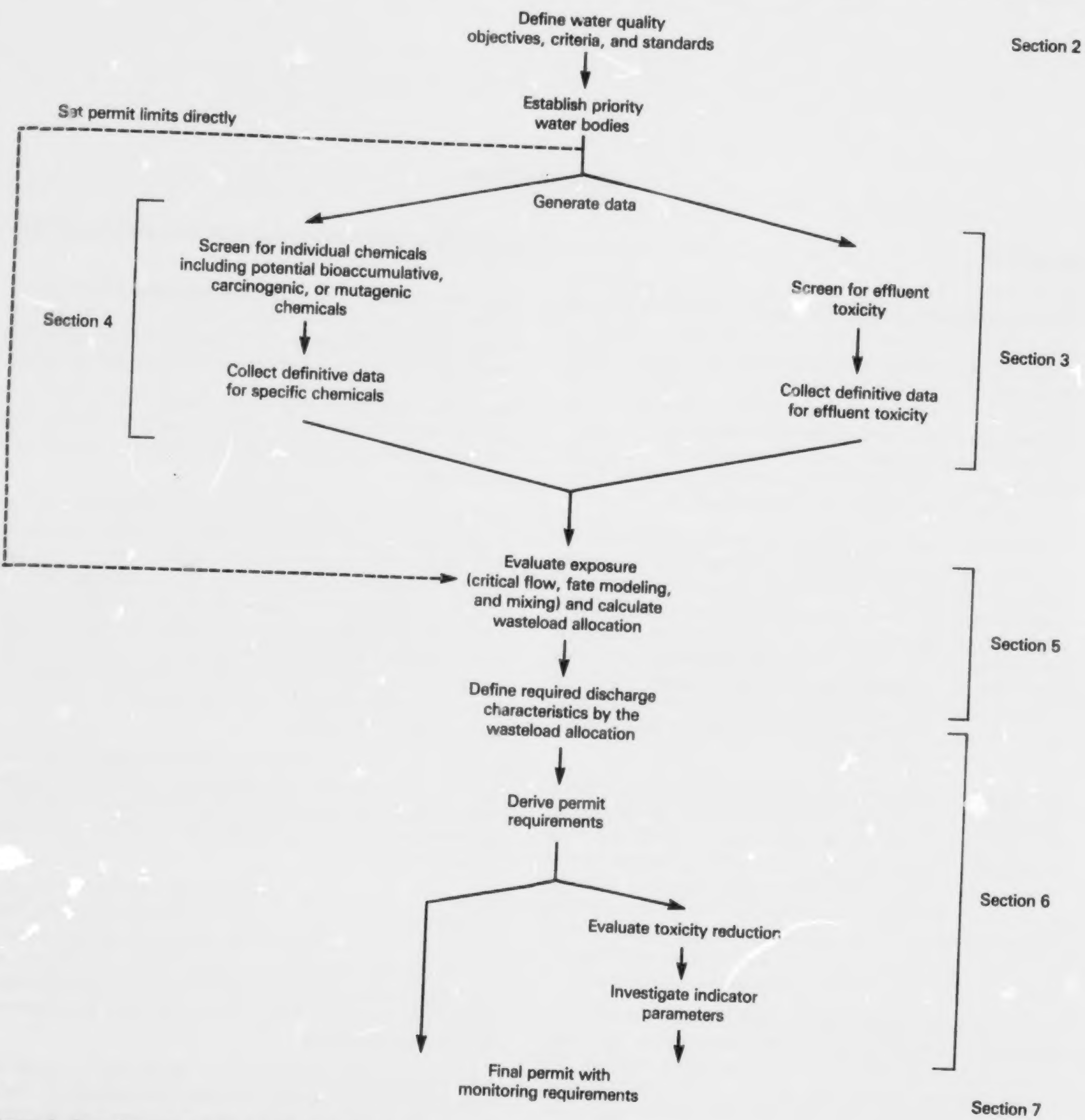

Define water quality objectives, criteria, and standards
Section 2
Establish priority water bodies
Set permit limits directly
Generate data
Screen for individual chemicals including potential bioaccumulative, carcinogenic, or mutagenic chemicals
Section 4
Collect definitive data for specific chemicals
Screen for effluent toxicity
Section 3
Collect definitive data for effluent toxicity
Evaluate exposure (critical flow, fate modeling, and mixing) and calculate wasteload allocation
Section 5
Define required discharge characteristics by the wasteload allocation
Derive permit requirements
Section 6
Evaluate toxicity reduction
Investigate indicator parameters
Final permit with monitoring requirements
Section 7

Figure 1. Overview of the water quality-based toxics control process.

Section 8, Case Example, illustrates how the recommended procedures from each preceding section are applied in a toxic water quality impact situation. A multiple source toxic impact site is described, and effluent characterization, exposure assessment, permitting, and monitoring requirements for the site are detailed step by step. The purpose of this example is to tie the recommendations contained in each section together to provide an overall picture of the water quality-based toxics control process.

There are five appendices to this document. Several of these are more detailed discussions of specific subjects referenced in the text of the document. Others provide data or examples of different techniques.

1. *Approaches to Water Quality-based Toxics Control*

Two approaches to the assessment and control of toxics are presented in this document: the whole-effluent approach and the chemical-specific approach.

The Whole-effluent Approach

The whole-effluent, or toxicity-based, approach to toxics control involves the use of toxicity tests to measure the toxicity of industrial and municipal wastewater discharges. A toxicity test is a relatively simple laboratory procedure. An effluent sample is collected and diluted in test chambers; usually the dilutions used are 100%, 30%, 10%, 3%, 1%, and a control. Often the receiving water is used to dilute the effluent because it more closely simulates effluent/receiving water interactions. Test organisms are placed in these test chambers for specified periods of time. At various points during the testing period, the number of organisms affected in each chamber is counted (the endpoint can be mortality, lower fecundity, reduced growth rates, etc.) and the lowest effluent concentration that causes that endpoint is calculated. This endpoint concentration becomes a quantified measure of the concentration that would cause instream impact if exceeded for a particular period of time. It is usually stated either as an LC50 (the effluent concentration at which 50% of the test organisms are killed) or a No Observed Effect Level or NOEL (the highest effluent concentration at which no unacceptable effect will occur even at continuous exposure).

The measurement of whole-effluent toxicity can then be used to limit the discharge of toxicants in an effluent. Toxicity itself is used as the effluent parameter. The toxicants creating that toxicity need not be specifically identified or controlled where the effluent's toxicity is limited. An analogy between effluent toxicity and biochemical oxygen demand (BOD) can be drawn. Both are measurements of a biological effect. Both can be quantified. In neither case are the causative agents of the biological effect specifically identified. Thus, whole-effluent toxicity is like BOD in that it is a useful parameter for reducing an undesirable effect caused by the discharge of a complex mixture of waste materials.

Permit limits can be expressed as LC50s or NOELs (e.g., the effluent shall exhibit an LC50 or NOEL of greater than 10%). The proper derivation of such toxicity-based permit limits is described in Section 6. Due to the procedures involved in deriving the toxicity limits and since toxicity involves an inverse relationship (the lower the LC50 or NOEL the higher the toxicity of the effluent), it is better to translate concentration-based toxicity measurements into toxic units (TUs). In this way, the problem of the inverse relationship is overcome and the permit limit derivation process is better served. A toxic unit is simply 100 divided by the toxicity measured:

$$\mathrm{TU} = \frac{100}{\mathrm{LC50\ or\ NOEL}}$$

(where LC50 or NOEL is expressed as
percent effluent in dilution water)

An effluent with a toxicity of 10% is an effluent containing 10 TUs. A very important aspect of toxic units is that there are two different types depending on whether acute or chronic toxicity is measured. The proper expression of toxic units is $\mathrm{TU_a}$ and $\mathrm{TU_c}$. $\mathrm{TU_a}$ is the measurement of acute toxicity units, and $\mathrm{TU_c}$ is a measurement of chronic toxicity units (see the glossary for a definition of these terms). They are not the same measurement and should not be used interchangeably.

Acute and chronic TUs make it easy to quantify acceptable instream toxicity. By setting numerical water quality criteria for acute and chronic toxicity, regulatory agencies can quantify narrative State standards such as the "no toxics in toxic amounts" standard common to all State water quality standards. The recommended numerical whole-effluent toxicity criteria are described in Section 2.

The Chemical-specific Approach

The chemical-specific approach to toxics control involves the use of laboratory-generated water quality criteria or State standards to limit specific toxicants directly. The toxicity analysis of those chemicals is done in a comprehensive testing program that, unlike whole-effluent testing, attempts to consider a range of toxic endpoints including human health impact and bioaccumulation. Once a criterion is developed, the number is applied, through an exposure analysis, as a permit limit to ensure that the level of that toxicant is not exceeded after discharge. Fate modeling of the toxicant to estimate its behavior after discharge can be an important step in establishing water quality-based permit limits.

Advantages and Disadvantages of the Approaches

The principal advantages of chemical-specific techniques are:

- Treatment systems are more easily designed to meet chemical requirements because more treatability data are available and treatment engineers and permit writers are more familiar with the procedures.
- The fate of a pollutant can be measured through modeling.
- Chemical analyses, in simple cases, can be less expensive than toxicity testing.
- Specific problem chemicals such as carcinogens or bioaccumulative chemicals can be directly limited.

The principal disadvantages of chemical-specific techniques are:

- All toxicants in complex wastewaters may not be known and, therefore, control requirements for all potential toxicants cannot be set.
- It is not always clear which is the toxic compound(s) in the mixture.
- It can become expensive to measure individual toxicants, particularly where many are present in the mixture. Organic chemicals can, in particular, be costly to measure.
- The bioavailability of the toxicants at the discharge site are not assessed, and the interactions between toxicants (e.g., additivity, antagonism) are not measured or accounted for.

The principal advantages of the whole-effluent approach are:

- The aggregate toxicity of all constituents in a complex effluent is measured, and toxic effect can be limited by limiting one parameter — effluent toxicity.
- The bioavailability of the toxic constituents is assessed, and the effects of interactions of constituents are measured.

The principal disadvantages of the whole-effluent approach are:

- Effluent toxicity treatability data are lacking, and treatment engineers and permit writers are unfamiliar with the procedures.
- Where there are chemical/physical conditions (pH changes, photolysis, etc.) present that act on toxicants in such a way as to "release" toxicity downstream, such toxicity may not be measured in the effluent.
- Properties of specific chemicals in complex effluents (such as bioaccumulation, carcinogenicity) are not assessed.

Since each approach can complement the other, this document has been written to integrate them. This document focuses on the whole-effluent approach as a toxics control mechanism because it is a relatively new process when compared to the chemical-specific approach.

Experience has shown that certain issues can cause unnecessary problems and delays in toxics control because of a misunderstanding of their relative importance or technical merit in a water quality-based permitting situation. This is particularly true for the whole-effluent toxicity testing approach. The following discussion focuses on issues involved in the application of this approach.

Issues Involved in the Toxicity-based Toxics Control Process

Correlation of Effluent Toxicity Measurements to Actual Instream Impact

EPA has devoted much attention to this issue. Considerable criticism has been directed at the use of effluent toxicity tests to assess adverse water quality impact because it had not been shown that levels of effluent toxicity measured in aquatic toxicity tests correlated directly to instream toxicity and corresponding ecosystem degradation.

Research conducted specifically to address this criticism, EPA's Complex Effluent Toxicity Testing Program, has shown that at each of the eight sites investigated to date (Lima, Ohio; Circleville, Ohio; Birmingham, Alabama; Baltimore, Maryland; Waterbury, Connecticut; Enid, Oklahoma; Charleston, West Virginia; and Wheeling, West Virginia), when exposure and potential chemical interactions are assessed, effluent toxicity can be correlated directly to impact in receiving waters [1-8]. This relationship is proving valid in both "unimpacted" sites (Circleville) and "impacted" sites (the other sites listed above).

There is a growing consensus among aquatic biologists that an evaluation of effluent toxicity, when adequately related to instream conditions, can give a valid assessment of receiving system impact [9].

This is also true in the application of chemical-specific water quality criteria for toxicants. Provided exposure is adequately assessed, measured instream concentrations of individual chemicals exceeding established safe concentrations can be considered to cause unacceptable impact to the receiving system.

Toxicity Test Method Precision

Considerable concern has been expressed about the precision and reproducibility of toxicity tests because it was believed that the tests themselves exhibited a large inherent variability. Research into this issue by various groups has shown that toxicity test methods, where properly followed, exhibit an acceptable range of variability [10-18]. This variability is similar to test method variabilities exhibited by accepted analytical procedures for individual chemicals.

On-site vs. Off-site Toxicity Testing

Comparisons of toxicity data between tests conducted on-site to tests conducted off-site on samples shipped to Environmental Research Laboratory (ERL)-Duluth via airfreight have shown little variation. For many effluents, on-site or off-site test data do not appear to be significantly different. The major consideration is practicality.

Flow-through toxicity tests require a continuously pumped sample. If it is determined that flow-through tests are needed, on-site testing may be mandatory.

Cost should be weighed against data needs to make this determination. If it is not considered important to the analysis of toxic impact, off-site testing (which is cheaper and can result in the generation of more data) is as acceptable as on-site testing.

Flow-through vs. Static and Renewal Toxicity Testing

A flow-through toxicity test is often conducted using a diluter system and a continuous feed of effluent and dilution water. A static toxicity test is conducted in test chambers, without a serial diluter delivery system, into which effluent and diluent are added manually. Only one test sample is added at the beginning of the test. A renewal toxicity test uses the same delivery system as that of a static test but the test solutions are changed on a pre-determined schedule.

There are several factors to consider in making the choice of test system: what type of toxicity is being measured (is the wastewater highly variable or not; continuous or intermittent discharge?), how much data need to be obtained (variable effluents may require more data), and expense. Each testing method measures effluent toxicity. However, on-line continuous flow-through testing can catch "peaks" of toxicity (should they occur during the testing period) in variable effluents whereas static or renewal tests have a lesser probability of doing so.

Cost is also a factor. Flow-through tests are more resource intensive and require complex delivery systems. Consequently, less data can be generated than with static or renewal testing. Where more data at less cost are desirable, static or renewal testing is probably more appropriate.

Variability

There are three important sources of variability in a water quality impact analysis:

- Effluent variability caused by changes in the composition of the effluent.
- Exposure variability caused by changes in flow rates of both effluent and receiving water. There are also variable receiving water parameters that are independent of flow, such as background toxicant levels, pH, suspended solids, hardness, and temperature, that can be important in assessing impact.
- Species sensitivity variability caused by the differences in response to toxicants between species.

Effluent Variability

Effluent variability can be handled by a proper sampling and testing procedure. Different sampling techniques will result in the measurement of different aspects of toxic impact. For example, composite samples tend to smooth out peaks of toxicity but do stand a better probability of sampling those periods of peak toxicity. Grab sampling will measure peak toxicity but only if the sample is taken during that period of peak toxicity. Since this period may be very short, there can be a low probability of catching that peak. The probability will increase by requiring analysis of more grab samples.

Sampling measurements should be tailored to that toxicity measurement of importance and the need to design testing that accounts for effluent variability. Section 3 describes recommendations for testing frequency designed to assess variable effluents. Appendix B details suggested sampling procedures.

Exposure Variability

Exposure variability is a complex factor that can be addressed in two ways. A simpler, more easily applied approach is to assume a steady state exposure condition (usually an estimate of presumed "worst case" exposure using a low critical receiving water flow and a typical effluent flow).

A second method is to attempt to estimate or actually measure the variable exposure state at the discharge site. This requires statistical analysis and some form of dynamic modeling. Section 5 of this document describes appropriate exposure assessment procedures.

Species Sensitivity Variability

Different species exhibit different sensitivities to toxicants. Often several orders of magnitude difference exist between the least sensitive and the most sensitive species when they are exposed to a particular toxicant. This range varies greatly and can be narrow or wide depending on the toxicant involved. Therefore, generalizations about sensitive and non-sensitive species when considering effluent toxicity are difficult to make.

Trout are considered sensitive organisms requiring high quality water. This generalization is almost universally accepted. When dealing with toxicity, however, this generalization may be inaccurate. Trout are very sensitive to oxygen depletion but may be relatively insensitive to certain toxicants. *Daphnia magna*, a very sensitive test species when exposed to many toxicants, is relatively insensitive to exposure to the pesticide endrin. Bluegills are very resistant to metals, particularly copper. Conversely, they are in the sensitive range of test species for organophosphate pesticides. Figures 1-1, 1-2, and 1-3 graphically show these differences in species sensitivities to hexavalent chromium, dieldrin, and an effluent from a publicly owned treatment works (POTW), respectively. The wide range between sensitivities for the different test species is shown. Comparing the figures shows that the fish and the invertebrates shift relative sensitivities to the toxicants. The fish are less sensitive to chromium but more sensitive to dieldrin. For the cladocerans, the reverse is true. A comparison of species sensitivities to individual toxicants has recently been published [19].

The primary goal in establishing a toxicity testing requirement for a discharger is to find a sensitive test species. The indigenous biota exposed to that effluent in the receiving water are likely to exhibit a lesser sensitivity to that effluent than the sensitive test species and will be protected so long as that test species' no effect level is not exceeded.

Since the measured toxicity of an effluent will be caused by unknown toxic constituents, the relative sensitivities of the test species will also be unknown. Therefore, *proper effluent toxicity analysis requires an assessment of a range of sensitivities of test species to that effluent.* A knowledge of the range is necessary so that the regulatory authority can identify a test species among the different species tested that is truly sensitive relative to the indigenous biota. The only way to assess the range of sensitivities is to test a number of different species. To determine how many species to test, cost must be balanced against reducing scientific uncertainty.

Analysis of species sensitivity ranges found in the national water quality criteria documents indicates that if tests are conducted on three particular species (*Daphnia magna*, *Pimephales promelas*, and *Lepomis macrochirus*), the most sensitive of the three will have an LC50 within one order of magnitude of the most sensitive of all species tested [20]. This was found to be true for 71 of the 73 priority pollutants tested with four or more species.

To best balance cost with toxicological certainty, *EPA recommends three species be considered the number of test species needed to eliminate a sufficient portion of the uncertainty for this factor.*

Often, regulatory agencies have been requiring testing on resident species under the assumption that such tests are needed to assess impact to local biota. However, standard test species, which are easier to test than resident species (based on availability, etc.) will be representative of expected impact on resident species in most cases so long as the range of sensitivities is assessed and a sensitive test species is identified. Resident species toxicity testing is not recommended unless it is required by State statute or some other binding factor. Such testing is more costly, more difficult to do, and subject to more variability (disease, age, etc.) than standardized testing.

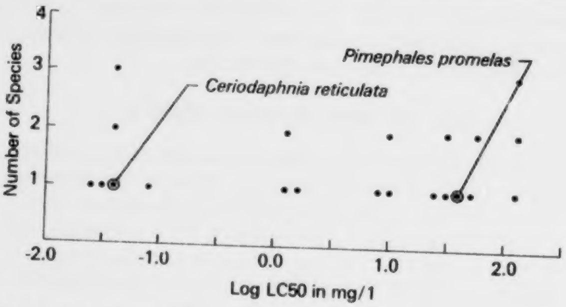


Figure 1-1. Log of LC50s of freshwater species exposed to hexavalent chromium.

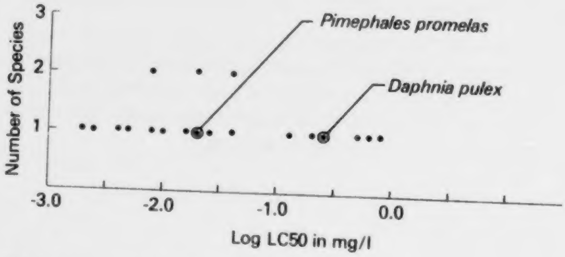


Figure 1-2. Log of LC50s of freshwater species exposed to dieldrin.

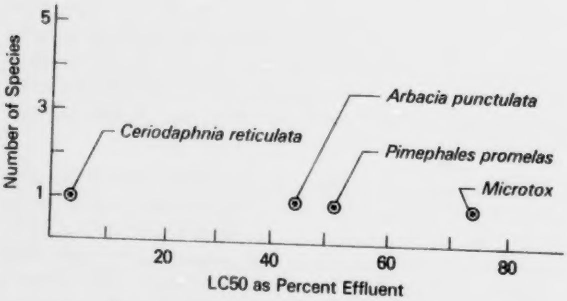


Figure 1-3. LC50s of freshwater and saltwater species exposed to a POTW effluent.

Acute-Chronic Ratio (ACR)

The ACR expresses the relationship between the concentration of an effluent or a toxicant causing acute toxicity to a species and the concentration of an effluent or a toxicant causing chronic toxicity to that same species (i.e., ACR = LC50/NOEL). It has commonly been used to extrapolate to a "chronic toxicity" concentration using an available acute toxicity data point. The most widely applied ACRs are 20 and 100; that is, chronic toxicity concentration is one-twentieth to one one-hundredth of the acute toxicity level. Commonly, 20 has been used for non-persistent toxicants and 100 has been used for persistent toxicants. These numbers have been used for both chemical-specific and whole-effluent approaches to impact assessment.

This parameter is a source of variability in assessing toxic water quality impact because the ACR varies both between species and, for any one species, between different toxicants. Therefore, to account for such differences, the regulatory authority must apply some ACR to an effluent when faced with a limited amount of toxicity data.

Research on acute and chronic toxicity for effluents shows that ACRs for whole effluents are more often around 10 [1-8]. When dealing with effluent toxicity, EPA recommends regulatory agencies use 10 as an ACR, in the absence of specific information.

This value can be used both to extrapolate to chronic concentrations from acute toxicity data and to set permit limits limiting chronic toxicity where chronic toxicity is not directly measured.

Of course, where acute and chronic toxicity data are available, the ACR can be directly calculated for that specific effluent. With this value, more cost-effective monitoring can be conducted using acute tests only. However, if the effluent's constituents vary, so will the ACR.

Behavior of Toxicants and Toxicity after Discharge

An understanding of the behavior of both single toxicants and whole-effluent toxicity after discharge may be necessary in the application of water quality-based toxics controls. Evaluating the combined effect of interacting toxic discharges may also be important in multiple discharge situations. When evaluating the instream behavior of toxicants and toxicity, certain factors become important.

- Persistence — How long, to what extent in terms of area, and at what level does effluent toxicity or the toxicity of a single toxicant persist after discharge? Is it reasonable to assume that the persistence of both individual toxic chemicals and effluent toxicity is conservative?
- Additivity, antagonism, and synergism — When toxicants or effluents with toxic properties mix in the receiving water, what is their combined toxic effect?

These factors are discussed below.

Persistence

As soon as an effluent mixes with receiving water, its properties begin to change. The rate of change of toxicity is a measure of the persistence. In most cases, the level of toxicity instream will either remain relatively constant (until further diluted by some additional source), or drop as decay processes (photodecomposition, microbial degradation) or compartmentalization processes (sediment deposition, volatilization) occur.

On-site toxicity testing has indicated that the toxicants causing toxicity measured at discharge sites tend to be persistent. There may be little near field degradation of measured effluent toxicity. Effluent toxicity does exhibit far field decay. Typical patterns of progressive downstream decreasing toxicity (similar to BOD decay) have been observed in a number of discharge situations [1-8].

Since mixing zones designated by State standards are usually restrictive enough to render toxicity assessments essentially near field situations, toxicity degradation may not be important in these permitting situations. Toxicity can be considered conservative (non-degrading) in these cases. However, if a permittee believes that his effluent's toxic effect is non-persistent, several measurement techniques can be employed to assess changes of toxicity after discharge.

- The toxicity test itself, when performed with dilution water from immediately upstream, is an analogue of the mixing processes taking place in the receiving water. Rapid chemical reactions that occur in the mixing zone can be expected to occur in the dilution procedure. The effects of varying physical/chemical conditions on persistence cannot be predicted from these results, however. Testing should be performed during critical low flow periods.
- Ambient toxicity testing, as detailed in Appendix C, measures the instream interactions of effluent and receiving water and can be used to assess persistence.
- Artificial "aging" of an effluent by holding the sample on the shelf for a specific period of time is another technique that can be used to estimate persistence.

One disadvantage of the chemical-specific approach is that the bioavailability of the toxicant after discharge is not measured. However, persistence can be modeled and the persistence of specific toxicants can be accounted for in making impact predictions and setting controls.

Additivity, Antagonism, and Synergism

ADDITIVITY

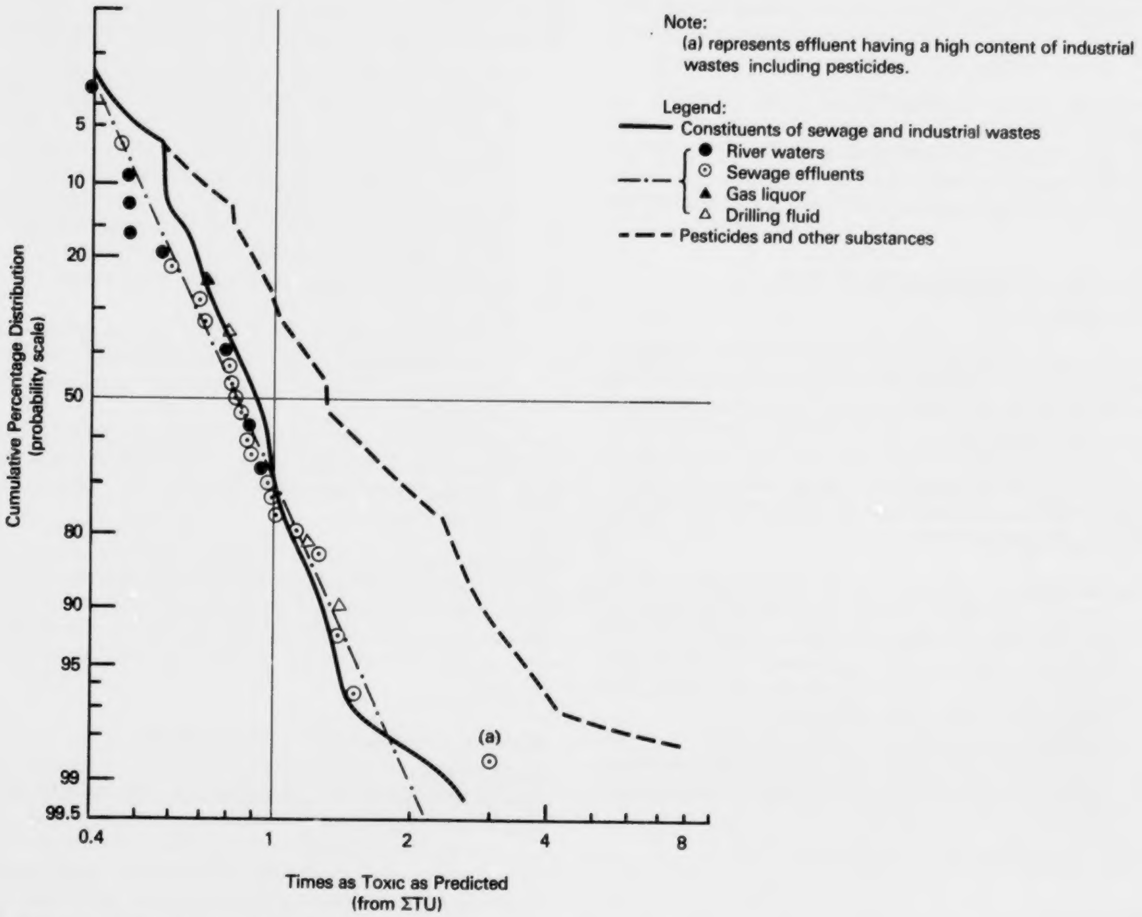
The issue of additivity can be stated as follows: if multiple effluents are discharged to a receiving water, is the toxicity additive in some way? Since each effluent is composed of individual toxic substances, the mixture of effluents in a receiving water (assuming conservative behavior for the moment) produces a mixture of these individual pollutants. The concentrations of each can be calculated from mass balance. Thus, the question of additivity of toxic effluents is the same as the question of additivity of the toxicity of mixtures of chemicals. An extensive review of this question is available [21]. The Chapter 11 summary section is relevant:

> Examination of available data using this [additivity] model shows that for mixtures of toxicants found in sewage and industrial effluents, the joint acutely lethal toxicity to fish and other aquatic organisms is close to that predicted assuming simple addition of the proportional contribution from each toxicant. The observed median value for the joint effect of these toxicants on fish is 0.95 of that predicted; the corresponding collective value for sewage effluents, river waters and a few industrial wastes, based on toxicity of their contituents, is 0.85, while that for pesticides is 1.3. The less than additive effect of commonly occurring toxicants in some mixtures may be partly attributable to small fractions of their respective LC50s having little or no additional effects.

Figure 1-4 illustrates Alabaster and Lloyd's data summary. Over 90% of the data, from the 5 to 95 percentile normalized for "relative" acute toxicity, are within a factor of two of confirming the additivity model. Thus, for acute toxicity at least, mixtures of effluents will probably exhibit additivity in terms of relative acute toxicity.

Little information appears to be available on the additivity of chronic toxicity. For the growth of fish, Alabaster and Lloyd conclude:

> On the other hand, in the few studies on the growth of fish, the joint effect of toxicants has been consistently less than additive which suggests that as concentrations of toxicants are reduced towards the levels of no effect, their potential for addition is also reduced. There appear to be no marked and consistent differences between the response of species to mixtures of toxicants.



Note:
(a) represents effluent having a high content of industrial wastes including pesticides.

Figure 1-4. Data summary on additivity from Alabaster and Lloyd [19].

Additivity appears to supply a reasonable estimate of the combined effects of individual acutely toxic agents. Toxicities approaching the NOEL do not appear additive, however. This suggests that additivity will produce an upper bound of the actual chronic toxicity.

ANTAGONISM

Cases in which one effluent ameliorated the toxicity of another effluent (antagonism) have been observed. If practical, a testing procedure designed to measure such interaction can be employed. A description of such a procedure is found in Section 3, under the heading Multiple Source Discharge Situations.

SYNERGISM

Under certain conditions, synergism, a greater-than-additive increase in toxicity upon mixing, can theoretically occur. However, field studies involving effluent toxicity and laboratory experiments with specific chemicals imply that synergism is an extremely rare phenomenon (it has not been observed during on-site effluent toxicity studies) and may not be an important factor in the toxicological assessment of effluents.

It should be noted that ambient toxicity tests can give some indication of the presence or absence of each of these factors.

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2. *Water Quality Standards*

The starting point for a water quality-based control program is the determination of the level of water quality adopted for the waterbody. The following discussion describes standards and provides guidance on narrative and numerical criteria that, when met, will achieve water quality standards. It should be noted that the water quality criteria recommendations presented in the TSD are based directly on the national water quality criteria (including averaging periods). These recommendations are subject to change if and when the Agency determines any changes, improvements, or updates are needed.

Water Quality Standards

Water quality standards are adopted by States (or, where necessary, promulgated by EPA) to define designated uses and ambient characteristics of receiving waters that must be maintained to allow those uses. These standards must be met in order to discharge wastewaters legally. Standards are composed of three parts: 1) an antidegradation statement; 2) designated uses for individual waterbodies; and 3) narrative and/or numerical criteria. Some criteria apply State-wide, whereas others are specific to particular designated uses or waterbodies.

In recent years EPA has sought to reinforce the central function of water quality standards and to emphasize the process for establishing and reviewing standards. The new water quality standards regulation (see 40 CFR 131; 48 FR 51400, November 8, 1983) and the Water Quality Standards Handbook [1] implement substantial changes in the standards program. To support certain standards actions, States must assess the attainability of uses (e.g., whether natural or irreversible man-made factors preclude achieving a designated use)[2]. States are also required to establish new criteria to control toxic materials and detail in their standards or planning documents how criteria will be implemented. Flexibility has been allowed in establishing chemical criteria different from the national criteria in order to account for local site-specific factors. Several steps have also been taken to better involve the public in decision making.

An important feature of the water quality standards regulation is an antidegradation requirement. If a designated use is currently being attained, the waterbody may not be classified for a less stringent use. For example, if a waterbody is being used for fishing and potable water supply, the waterbody must be classified for those or more stringent uses regardless of the current classification or development pressures. Also, if water quality is better than necessary to maintain aquatic and recreation uses, that level of water quality must be maintained unless the State meets the conditions discussed in the Water Quality Standards Regulation.

Mixing Zones

Many State water quality standards allow a zone of mixing in which less stringent criteria apply than apply to the rest of the waterbody. The rationale is that a small area of degradation can exist without causing adverse effects to the overall water body. The criteria that apply to mixing zones vary from State to State. In some States there are explicit requirements for water quality within mixing zones (such as no acute toxicity, floating materials, or deposit-forming solids). In other States there are no requirements or the requirements are ambiguous. EPA's policy is described in the Handbook [1] and discussed in detail in Section 5.

The allowable size of mixing zones also varies by State. Most States specify that the zone must not be as wide as the stream in order to allow a zone of passage for fish. Very few States specify the allowable length. Usually, the size of the mixing zone is determined on a case-by-case basis taking into account the critical resource areas that need to be protected.

It is important to note that mixing zones should be evaluated and used for regulation in cases where mixing is not complete within a short distance of the outfall. In the majority of cases involving conventional pollutants, mixing has been assumed to be complete since the impact of these pollutants occurs downstream. The regulatory authority has conducted the evaluation of the discharge and calculation of permit limits based on effluent dilution in the full stream flow. Evaluation and control based on a mixing zone has usually been limited to situations where mixing is known to be poor (e.g., shore-hugging plumes and discharges to large rivers, lakes, and estuaries). However, if mixing is assumed to be rapid and complete when it is not, a toxic discharge that appears to meet standards may cause zones of chronic toxicity that extend for miles. Therefore, regulatory agencies should carefully evaluate mixing. Methods to evaluate dispersion and set mixing zones are included in Section 5.

Magnitude, Duration, and Frequency

Criteria are specifications of water quality designed to ensure protection of the designated use. EPA criteria are developed as national recommendations to assist States in developing their standards and to assist in interpreting narrative standards. EPA criteria consist of three components:

- Magnitude — How much of a pollutant (or pollutant parameter such as toxicity), expressed as a concentration, is allowable.
- Duration — The period of time over which the instream concentration is averaged for comparison with criteria concentrations. This specification limits the duration of concentrations above the criteria.
- Frequency — How often criteria can be exceeded without unacceptably affecting the community.

For each parameter, specifications for these components are developed for two levels of effect. These levels of effect are typically acute effects and chronic effects (criteria for dissolved oxygen are further subdivided). Thus, a criterion will consist of at least six specifications: recommended magnitude, duration, and frequency values for both acute and chronic levels of protection.

Magnitude for Single Chemicals

Water quality criteria for aquatic life contain two expressions of allowable magnitude: a criterion maximum concentration (CMC) to protect against acute (short term) effects; and a criterion continuous concentration (CCC) to protect against chronic (long term) effects. The two concentrations for the priority pollutants are presented in EPA's criteria documents. These criteria generally apply after mixing.

Most State standards include numerical criteria for only a few individual toxic chemicals. Generally, evaluation and control of toxic pollutants is based on maintenance of the designated use and the narrative criterion prohibiting toxic substances in toxic amounts.

The adverse effects of concern will depend on the designated use and the chemical. Human health hazards, bioaccumulation of chemicals in aquatic organisms, toxicity to these organisms, and aesthetic factors may be important. Available information on the toxic effects of the chemical is used when standards do not include specific numerical criteria. Such information can include EPA criteria documents, published literature reports, or studies conducted by the discharger.

Water quality-based controls may be based directly on the State's technical determination of what concentration of a specific pollutant meets the State's narrative "free from" toxics criterion. Although the new water quality standards regulation requires that the State's process for implementing its narrative criterion be described in the State standards, there is no requirement that this concentration be adopted as a numerical criterion in State water quality standards prior to use in developing water quality-based controls.

The Water Quality Standards Regulations allow States to develop numerical criteria or modify EPA's recommended criteria to account for site-specific factors. In cases where additional toxicological data are needed to modify or develop criteria, the discharger may be required to generate the data. Guidance on modifying national criteria is found in the Handbook [1]. When a criterion must be developed for a chemical for which a national criterion has not been established, the regulatory authority should refer to the Guidelines for Deriving Criteria for Aquatic Life and Human Health (see 45 FR 79341: November 28, 1980). Revisions to the guideline for deriving aquatic life criteria have been finalized (see 50 FR 30784, July 29, 1985).

Magnitude for Whole-effluent Toxicity

Criteria for toxicity in current State standards range from the narrative prohibition (e.g., no discharge of toxic chemicals in toxic amounts) to detailed requirements that specify the test species and the allowable toxicity level. State standards often include the application factors (to derive a "safe" toxicity value from an acute toxicity value) that were recommended in the 1972 "Blue Book" guidance (i.e., 0.05 for nonpersistent wastes and 0.01 for persistent wastes)[3].

EPA's recommended magnitudes for whole effluent toxicity are as follows. Again, two expressions of allowable magnitude are used: a CMC to protect against acute (short term) effects; and a CCC to protect against chronic (long term) effects.

For acute protection, *the CMC should not exceed 0.3 acute toxic unit (TU_a)* as measured by the most sensitive of at least three test species. As explained in the previous section, the selection of test species is not critical provided species from ecologically diverse taxa are used (e.g., a fish, an invertebrate, and a plant). The factor of 0.3 is used to adjust the typical LC50 endpoint of an acute toxicity test (50% mortality) to an LC1 value (virtually no mortality). Specifically, a factor of 0.3 was found to include 91% of observed LC50 to LC1 ratios in 496 effluent toxicity tests as illustrated in Figure 2-1. This figure presents effluent toxicity data from many years of toxicity testing of both industrial and municipal effluents by the Environmental Services Division, U.S. EPA Region IV, Athens, Georgia.

For chronic protection, *the CCC should not exceed 1.0 chronic toxic unit (TU_c)* to the most sensitive of at least three test species. The selection of test organisms is as described above.

For both the acute and chronic magnitude, if fewer than three species are tested, the test results are divided by

10 to account for species sensitivity differences (see Section 1 for discussion). Thus, if only two species are used to measure the toxicity of the receiving water or effluent discharges, the CCC would be 0.1 TU_c.

The following example illustrates the use of these magnitudes for toxicity. Assume that an effluent discharge is the only source of toxicity to a water body and that the available dilution is 30 to 1 (dilution factor = 30). The basic relation between criteria concentrations, dilution, and allowable effluent quality can be stated as follows:

$$\text{allowable effluent concentration} \times 1/\text{dilution factor} \leq \text{criteria concentrations}$$

Using this equation with the criteria, the effluent acute toxicity should be less or equal to nine TU_a (or $LC50 \geq 11\%$ effluent) and the effluent chronic toxicity should be less than 30 TU_c (or NOEL $\geq$ 3.3% effluent).

As with single chemicals, water quality-based controls may be based directly on the State's technical determination of what level of toxicity meets the State narrative toxics criterion. Again, the process should be described in the State standards but this level of toxicity need not be adopted as a numerical criterion prior to use.

Duration

The quality of an ambient water typically varies in response to variations of effluent quality, stream flow, and other factors. Organisms in the receiving water are not experiencing constant, steady exposure but rather are experiencing fluctuating exposure. Short periods of exposure to high concentrations of toxicants can have adverse effects. Thus, EPA's criteria restrict the maximum time period over which exposure is to be averaged, thereby limiting the magnitude and duration of exposure.

For the CMC, the maximum period in which to average the exposure is one hour. Thus, the one-hour average exposure must be less than the CMC to protect against acute effect. In practice, one-*day* periods are the shortest periods for which wasteload allocation (WLA) modelers and enforcement personnel have adequate data. Compliance with the duration criterion can be ensured by paying particular attention to short term effluent variability and requiring measures to control variability (e.g., installation of equalization basins) when needed.

For the CCC, the maximum period in which to average the exposure is four days. Some pollutants may have magnitude criteria that correspond to different periods. The toxicity criterion for chronic effects is specified as a four-day average. Thus, the four-day average exposure must be equal to or less than the CCC.

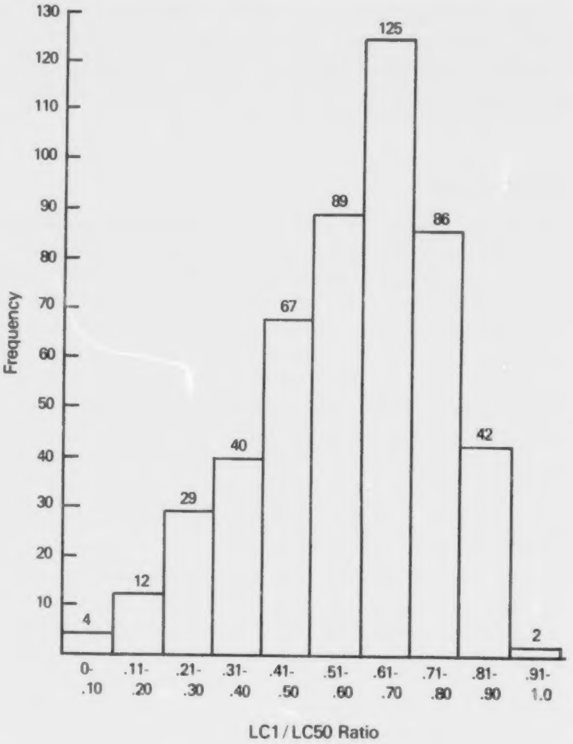

Figure 2-1. LC1 to LC50 ratios for effluent toxicity tests.

These specifications for duration apply to both chemical-specific and whole-effluent toxicity criteria. Many people have erroneously assumed that, because many chronic toxicity tests are 28 or 30 days in length, the CCC was meant to be used as a 30-day average. However, the duration of a toxicity test has nothing to do with the critical period of exposure to concentrations greater than the criteria. Many chronic toxicity tests are of a one-year or longer duration, yet this does not lead to the establishment of an averaging period of one year's duration. Obviously, if a one-year averaging period were used, the CCC could theoretically be exceeded for six months, a duration more than long enough to cause an unacceptable chronic effect in a waterbody.

The toxicity tests used to establish the national criteria numbers are conducted using steady state exposure techniques. They do not account for fluctuating exposures such as those experienced by the instream biota due to effluent variability. What is pertinent is that as the period of averaging increases, so too does the period of time the exposure concentrations can be above the criterion concentration without exceeding the average. The question is, how long can the exposure concentration be above the criterion concentration without significantly affecting the endpoint of the test (e.g., survival, growth, or reproduction)? This is the significant consideration involved in setting duration criteria. The period of time above the criteria concentration must be limited. The four-day duration recommended by EPA specifies the period of time over which to average exposure. This duration time was derived to limit the duration above the CCC.

Frequency

The application of criteria involves predictive modeling of ambient conditions based on the assimilative capacity of the receiving water and sources of pollution. One essential element in this modeling is an assessment of the projected frequency with which criteria will be exceeded. There are two reasons why this frequency is a factor in water quality criteria. First, it is statistically impossible to project that criteria will never be exceeded (unless pollution sources are removed entirely). Second, ecological communities are able to recover from stresses. Since WLA modeling requires a specification for frequency, it is important that ecologically sound recommendations be provided.

The frequency with which criteria are allowed to be exceeded depends on site-specific factors. To implement the criteria, site specific factors as described in Appendix D must be taken into account. EPA's recommendations for frequency are *once in three years for both the CMC and the CCC.* These recommendations apply to both chemical-specific and whole-effluent approaches.

As explained in Appendix D, field studies indicate that many discharge situations are affected both by predictable and measurable discharges of toxicants and by unpredictable spills of toxic substances. In most cases the dischargers were unaware that spills were occurring. These spills are a second source of stress for the community and decrease recovery potential. An aggressive program to minimize, contain, and treat spills should be in place at any plant where the potential for spills exists before using the three-year frequency criteria.

Detailed guidance on derivation of design flows for steady state modeling will be provided by EPA. Until this detailed guidance is issued, the 1Q10 and 7Q10 should be used as an interim design flow for stressed systems and the 1Q5 and 7Q5 should be used for unstressed systems, where conditions discussed in Appendix D are met and when dynamic modeling is not undertaken.

References

1. U.S. Environmental Protection Agency. 1984. Water Quality Standards Handbook. Office of Water Regulations and Standards (WH-585), Washington, D.C.
2. U.S. Environmental Protection Agency. 1984. Technical Support Document for Conducting Use Attainability Studies. Office of Water Regulations and Standards (WH-585), Washington, D.C.
3. U.S. Environmental Protection Agency. 1972. Water Quality Criteria. EPA-R3-73-033.

3. *Effluent Characterization*

The purpose of effluent characterization is to provide the data needed to determine whether or not the discharge of toxic materials causes adverse impact to the biota of the receiving system.

This section of the Technical Support Document discusses specific measurement techniques and overall analytical procedures to be followed to achieve this objective. The specific procedures employed at any one site should be tailored to that site to account for its unique characteristics. Aspects of the procedures discussed below are based on conclusions reached at the Fifth Pellston Environmental Workshop held in August, 1982, at Cody, Wyoming [1].

Two general measurement techniques can be employed in effluent characterization; whole-effluent toxicity analysis and chemical-specific analysis. Whole-effluent toxicity testing is most appropriate in the following situations:

- Where an effluent's constituents are not completely known or where a complex mixture of potentially additive, antagonistic, or synergistic toxic pollutants are discharged.
- Where more than one discharger is located in a specific area and the potential exists for effluent mixing and additive toxic effect.
- Where chemical-specific evaluation is impractical due to a lack of information about the toxic effects of a chemical or a lack of the resources required to model the chemical(s) present.

Chemical-specific analysis is most appropriate in these discharge situations:

- Where an effluent contains only one or several well quantified pollutants for which toxicological data are available or can be generated, or where one or more exists as a large percentage of the waste flow.
- Where bioaccumulative, carcinogenic, teratogenic, or mutagenic chemicals are suspected in the effluent and specific limits for each of these chemicals may be required.
- Where potentially toxic constituents of an effluent may be masked, but cause delayed toxicity when acted on by downstream processes.

EPA's recently issued policy (Appendix A) specifically suggests the use of an integrated strategy for toxics control. There will be discharge situations where both toxicity analysis and chemical-specific analysis should be used. An example of such a situation would be where a known toxicant is discharged at very high levels in a complex effluent known to cause a toxic effect. A limit would be set for that toxicant specifically. For example, suppose ammonia is discharged by a publicly owned treatment works (POTW) at toxic levels. Following the policy, a limit for ammonia would be set independently to reduce these levels. An additional whole-effluent toxicity limit would be set to control the toxicity of the other toxicants in the effluent when it is suspected continued toxicity will exist after the limit on ammonia is implemented. An example of such an approach is described in Section 8.

Toxic impact is characterized by measuring effluent toxicity or concentrations of individual chemicals and comparing these measurements to the exposure concentrations of that effluent or individual chemical in the receiving water. The potential for impact is minimal where the exposure concentration is less than the toxic effect concentration and maximal where the exposure concentration approaches or exceeds the toxic effect concentration. A simple equation can be used to illustrate this:

$$\text{IWC} \leq \text{NOEL}$$

This is stated as "the Instream Waste Concentration (IWC) of an effluent or a toxicant must be less than or equal to the No Observed Effect Level (NOEL)." A NOEL is the highest concentration of a toxicant or an effluent that, if maintained over time, is expected to cause no biological impact. An IWC is the concentration of a toxicant or an effluent in the receiving water after mixing. The most appropriate or allowable mixing is determined by the regulatory authority. For single chemicals, the NOEL is the chronic water quality criterion number. For whole effluents, the NOEL is the measured or estimated chronic toxicity concentration as determined by the appropriate toxicity test on a sensitive test species.

This section provides recommendations for generating the data to determine the NOEL. Section 5, Exposure Assessment, provides recommendations for more detailed assessment of the IWC and the calculation of wasteload allocations for establishing permit limits. Section 6 describes the actual calculation of permit limits.

Before presenting EPA's effluent characterization testing recommendations, an important principle should be discussed. *It is not necessary to generate any effluent* *toxicity data to set toxicity-based permit limits.* Regulatory authorities do not need to require toxicity tests of a permittee nor do they need to generate the data themselves to limit effluent toxicity. Just as a water quality-based permit limit can be set for a single toxicant (based on the available dilution and the water quality criterion or State standard) in the absence of effluent data, so too can a toxicity limit be set in the absence of effluent toxicity data. Again, the available dilution and the water quality criterion for toxicity, as described in Section 2, can be used to establish a toxicity-based permit limit. The basic relationship is expressed as:

$$\text{allowable effluent toxicity} \times 1/\text{dilution factor} \le \text{criterion concentration}$$

where effluent toxicity and the criterion are expressed in toxic units and the dilution factor is the factor by which the effluent is diluted in the receiving water (e.g., one gallon effluent into nine gallons receiving water yields a dilution factor of 10). Since both the criterion for toxicity and the dilution factor are known, the value for allowable effluent toxicity can be calculated as follows:

$$\text{allowable effluent toxicity} \le \text{criterion} \times \text{dilution factor}$$

For example, suppose a State suspects toxicity impact by a discharger. The criterion used is 1.0 chronic toxic units (TU_c) at the edge of the mixing zone. The dilution factor is based on the flow immediately above the outfall at the 7Q10 of the river (complete mixing is assumed) or 10 cubic feet per second (cfs). The effluent flow is 0.5 cfs. Using the equation above:

$$\text{allowable effluent toxicity} \le 1.0\ TU_c \times \frac{10 + 0.5}{0.5}$$

$$\le 21\ TU_c$$

The allowable toxicity is $\le 21\ TU_c$ or NOEL $\ge 4.8\%$. In this case, the State can set a permit limit of 21 TU_c without requiring effluent toxicity testing. This does not imply, however, that toxicity testing is not necessary to ensure compliance with the permit limit. Toxicity monitoring would be used to determine if the discharger was meeting this limit.

For multiple sources, two additional considerations are involved. First, toxicity additivity must be assumed. Second, the flows must be summed and allowable toxicity allotted to the individual sources using some option for establishing toxicity wasteload allocations (WLAs) (see Section 5). The relationship for multiple sources is expressed as:

$$\frac{\Sigma\,(Q_e T_e)}{Q_s} \le \text{criterion}$$

where: Q_e = effluent flow
T_e = effluent toxicity
Q_s = stream flow at the point of reference (below the last source)

An example of the use of this equation is in Section 8, Case Example.

Of course, there are advantages and disadvantages to whichever route is taken (i.e., setting limits directly or engaging in effluent characterization). For setting permit limits directly without effluent characterization, the advantages and disadvantages are as follows:

- Advantages
  - The cost of data generation prior to permit issuance is eliminated.
  - The permit limit is quickly derived and permit issuance is not delayed.
- Disadvantages
  - Uncertainty as to whether or not a real toxicity problem exists because no data are available.
  - Exposure must be simplified so that only a steady state exposure condition can be used. This can be overprotective or underprotective depending on the design flow utilized.
  - The permittee may object to the limit since no data are available to show toxicity is an actual problem.

Where the regulatory authority decides to proceed directly to effluent characterization, the advantages and disadvantages are as follows:

- Advantages
  - Will indicate whether or not a real toxics problem exists.
  - Can generate the data needed to assess exposure in a more sophisticated manner. A sensitive test species is identified and variability is assessed.
  - The permittee is less likely to object to subsequent limits because a sound basis for the requirement is established.
- Disadvantages
  - The cost of testing can be high.
  - The testing and analysis could delay permit insurance.

Given these advantages and disadvantages, the regulatory authority must decide which course to take on a site-specific basis.

Tiered Testing

The most cost-effective way to generate the data necessary to assess impact is to follow a "tiered" testing procedure. The concept of tiered testing is similar to that used under the Toxic Substances Control Act (TSCA) when testing a new chemical product for potential hazard. Testing begins with short, inexpensive screening tests and progresses to more and more definitive data generation techniques.

The tiered approach integrates effluent effect measurements and simplified exposure assumptions into various stages of analysis and arrives at one of three decisions: 1) apparently safe because of a large margin between exposure (IWC) and effects (NOEL),

2) unacceptable hazard because the exposure concentration exceeds the NOEL, or 3) additional data are needed to resolve more clearly the marginal difference between exposure and effects. This exposure/effect level comparison enables a regulatory authority to determine where no further analysis is needed (apparently safe), where limit derivation should begin (unacceptable hazard), or whether there is a need for more complete effluent characterization (additional data needed).

Uncertainty Associated with Data Bases

All effects testing and exposure assessment parameters, for both effluent toxicity and individual chemicals, have some degree of uncertainty associated with them. The more limited the test data available, the larger the uncertainty factor becomes.

The least amount of uncertainty exists for an effluent's impact on the receiving water when a complete data base is available on acute and chronic toxicity on many indigenous species, when there is a clear understanding of ecosystem species composition and functional processes, and when actual measured exposure concentrations are available for all chemicals during seasonal changes and dilution capacities. The uncertainty associated with this ideal situation would be minimal. No manipulation of the data using adjustment factors or safety factors would be needed. Obviously, resource constraints generally prohibit the development of such ideal data bases at NPDES discharge sites.

The current method for calculating a national water quality critierion for a specific toxicant utilizes laboratory acute and chronic testing on a variety of species to arrive at the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC). These concentrations, if not exceeded, are expected to protect aquatic life (see 50 FR 30784, July 29, 1985) and thus have uncertainty factors of one. Uncertainty is minimized when chemical-specific limits are applied because a large amount of data has been generated on each toxicant.

For the purpose of evaluating which effluents need additional testing, each tier of data should be judged in light of the uncertainty of the data through the use of uncertainty factors. Table 3-1 lists recommended uncertainty factors. Uncertainty factors are factors that can be used to generate a specific level of uncertainty against which toxic effect concentrations/ambient exposure ratios are compared. For instance, a limited data base of a few acute toxicity tests on one species could require a level of uncertainty of 1,000 (derived from three uncertainty factors): 10X for species sensitivity, 10X for variability in effluent toxicity, and 10X for the acute-chronic ratio (ACR). (The ACR is a conversion factor, not an uncertainty factor. For simplicity, it is used in this context as an uncertainty factor.)

Table 3-1. Uncertainty Factors

Uncertainty Source	Uncertainty Factor	Rationale	Data Source
Effluent Variability	10	Range in effluent toxicity not observed to exceed one order of magnitude	[2a, 3]
	100	Heavy metals observed to vary two orders of magnitude in POTW effluents	[4]
Species Sensitivity	10	Acute or chronic tests using one vertebrate and one invertebrate	[2b]
Acute/Chronic Ratio	10	Value observed from on-site whole effluent toxicity testing	[5, 6, 7, 8, 9, 10]

To illustrate the use of uncertainty factors, suppose a discharger exhibits potential toxicity in a screening test. Further testing is required. The discharger conducts an acute toxicity test on a fish and measures an LC50 of 40%. The discharger's IWC is calculated by dividing the mean effluent flow, 3.0 cfs, by the 7Q10 of the river, which is 10,000 cfs. A level of uncertainty of 1,000 is calculated using the three applicable uncertainty factors; [10 (species sensitivity) × 10 (effluent variability) × 10 (ACR)]. To decide whether or not further testing is warranted, the regulatory agency compares toxicity and exposure to the level of uncertainty:

$$\frac{\text{LC50 (in percent)}}{\text{IWC}^{*}} > \text{level of uncertainty}$$

$$\frac{\text{LC50 (40\%)}}{\dfrac{3\text{ cfs}}{10{,}000\text{ cfs} + 3\text{ cfs}} \times 100} = 1{,}334 > 1{,}000$$

In this case, the ratio is greater than the calculated level of uncertainty (1,000). This discharger is given a low priority for further assessment. However, if the LC50 was 4% for this discharger, the ratio would not be greater than the level of uncertainty and further analysis would be warranted:

$$\frac{\text{LC50 (4\%)}}{\dfrac{3\text{ cfs}}{10{,}000\text{ cfs} + 3\text{ cfs}} \times 100} = 133 < 1{,}000$$

EPA recommends the concept of levels of uncertainty be used in judging the adequacy of a site-specific data for evaluating impact and calculating effluent requirements. These factors are perhaps more appropriate for the whole-effluent approach to toxic impact assessment. The species sensitivity and ACR uncertainty factors have been eliminated in the single chemical impact assessment through the criteria development process. Effluent variability can be considered the major uncertainty factor for chemical-specific evaluations.

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*The IWC can be calculated in two ways depending on the source of the facility's water supply. Use IWC = Q_w/Q_r where the source is the receiving water. Use IWC = Q_w/Q_w+Q_r where the source is not the receiving water (Q_w = wastestream flow; Q_r = receiving water flow.

Recommendations for conducting tiered toxicity tests are presented in Box 3-1.

Box 3-1
TESTING RECOMMENDATIONS

EPA recommends a two-tiered approach be used in toxic impact assessment. The first tier, SCREENING, is conducted to separate individual dischargers into those with potential water quality standards violations requiring further analysis and those where such violations are improbable. The second tier, DEFINITIVE DATA GENERATION, is conducted to generate sufficient data to quantify violations of water quality standards so that appropriate regulatory action can be taken to eliminate that violation. This tier of testing can be subdivided into progressively more detailed analysis. Figure 3-1 shows a schematic representation of the toxicity data generation process. Some methods for toxicity testing are listed in Table 3-2.

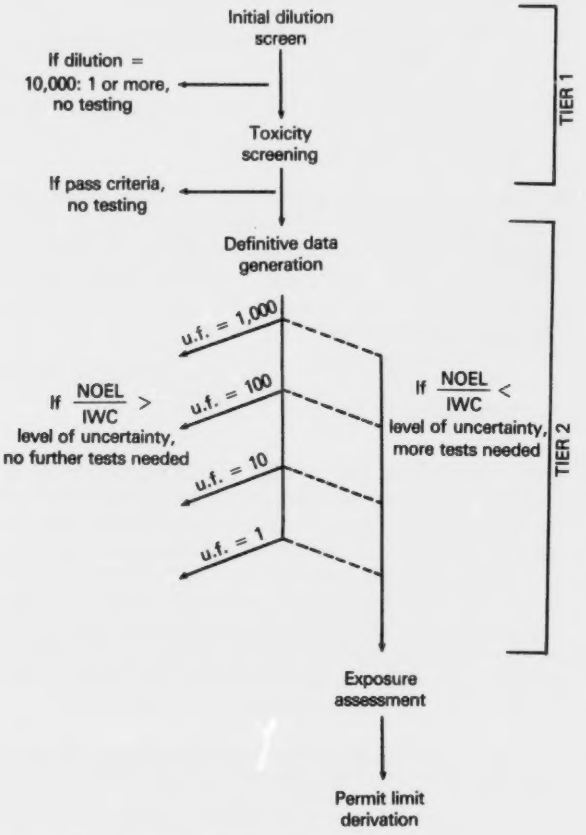
Initial dilution screen
If dilution = 10,000: 1 or more, no testing
Toxicity screening
If pass criteria, no testing
TIER 1
Definitive data generation
u.f. = 1,000
u.f. = 100
u.f. = 10
u.f. = 1
If NOEL/IWC > level of uncertainty, no further tests needed
If NOEL/IWC < level of uncertainty, more tests needed
TIER 2
Exposure assessment
Permit limit derivation

Figure 3-1. Data generation: tiered testing.

Table 3-2. Methods Manuals for Toxicity Testing

1. General Listing of Available Toxicity Tests

U.S. Environmental Protection Agency. 1982. Environmental Effects Test Guidelines. Office of Toxic Substances, Washington, D.C. EPA 560/6-82-002.

U.S. Environmental Protection Agency. 1982. Pesticide Assessment Guidelines. Office of Pesticide Programs. Washington, DC. EPA 540/9-82-018 through 028.

2. Acute Toxicity Tests

Peltier, W. and C. I. Weber, 1985. Methods for Measuring the Acute Toxicity of Effluents to Aquatic Organisms, Third Edition. Office of Research and Development, Cincinnati, OH EPA-600/4-85-013.

3. Chronic and Subchronic Toxicity Tests

Mount, D.I., and T.J. Norberg. 1984. A seven-day life cycle cladoceran toxicity test. Env. Tox. Chem. 3(3) p. 425-434.

Horning, W., and C.I. Weber, ed. 1984. Methods for measuring the chronic toxicity of effluents to aquatic organisms. Environmental Monitoring and Support Laboratory, Cincinnati, OH (in preparation). EPA-600/4-85-014.

Norberg, T.J., and D.I. Mount. 1985. A new subchronic fathead minnow (*Pimephales promelas*) Toxicity Test. U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, MN. (in Press).

Birge, W., and J. Black. 1981. In-situ acute/chronic toxicological monitoring of industrial effluents for the NPDES Biomonitoring Program using fish and amphibian embryo-larval stages as test organisms. U.S. Environmental Protection Agency, Office of Water Enforcement and Permits, Washington, D.C. OWEP-82-001.

4. Other Toxicity Tests

Bulich, A.A. 1982. Microtox System Operating Manual. Beckman Publication No. 015-555879. Beckman Instruments, Inc. Carlsbad, CA.

Tier 1. Screening

The goal of screening is to identify, with the most cost-effective method available, potential water quality impact situations and to avoid collecting unnecessary data in cases in which impact will probably be minimal. After conducting a screening analysis, cases should fall into two categories: those cases in which it is improbable that water quality is being adversely impacted, and those cases in which water quality is more likely to be impacted. In this first tier, false positives are more desirable than false negatives. Therefore, the assumptions and decision triggers used in evaluating the data should be conservative (i.e., stringent).

Determining which NPDES dischargers are candidates for water quality-based toxics control is of obvious concern. It is difficult to give recommendations as to which dischargers out of the active 65,000 NPDES permittees should conduct toxicity testing. This is a regulatory decision which must be made on a site-specific basis by State and regional regulatory authorities. Some factors pertinent in this decision are listed below.

- Dilution — Toxic impact is directly related to available dilution. The lower the available dilution, the higher the potential for toxic effect. One recommendation is if concentrations of instream effluents reach 1% or higher during critical low flow periods, those effluents should be given high priority for toxicity assessment. Assessment of the amount of dilution available on a national basis at annual low flow and at 7Q10 (Figure 3-2) shows that a majority of NPDES dischargers discharge to areas of low available dilution during certain critical periods.

- Type of industry — Although this factor should be carefully assessed, some generalizations can be made. The primary industrial categories should be of principal concern. Secondary industrial categories have less potential for toxicity than primary industries. Within the primary industrial categories no generalizations should be made. Toxicity problems associated with types of industries become site specific.
- The type and volume of industrial input (for POTWs) — POTWs with large loadings from indirect dischargers (particularly primary industries) are high priority candidates for toxicity testing. However, absence of industrial input is no guarantee of absence of toxicity problems.
- Existing data on toxic pollutants — Discharge monitoring reports (DMRs) and application form 2C data provide some indication of the presence of toxicants. Again, careful assessment is required. The presence or absence of the 126 priority pollutants is not an absolute indication of the presence or absence of toxicity. There are thousands of toxicants that may be causing effluent toxicity, not just 126. Also, low levels of several toxicants combined can cause toxicity where individually they would not.
- History of toxic impact and compliance problems — Regulatory authorities are usually well aware of particular problem areas. Such discharge sites would be priority candidates for toxicity analysis.

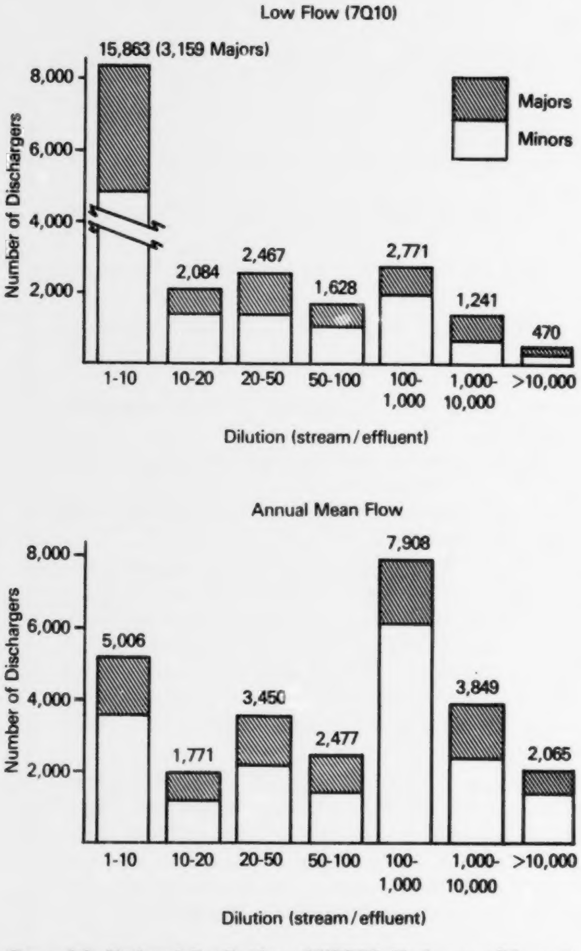

Figure 3-2. National distribution of NPDES dilution conditions at 7Q10 and at annual mean flow.

Given these factors, priorities for toxicity analysis can be set. Combinations of factors, such as low available dilution, poor compliance record, and clustered industrial and municipal discharges are obvious high priorities.

Where cases of water quality standards violations are already documented, it may be appropriate to skip the screening tier and go to the next tier. This is perhaps more applicable to situations where the chemical-specific approach is to be used because regulatory agencies usually have more chemical-specific data available. Effluent toxicity data are mostly unavailable for NPDES dischargers. For more complex or more toxicologically unknown discharge situations, data generation for whole-effluent toxicity screening will probably be needed to characterize the site fully.

The decision triggers associated with the tiers are obviously directly related to how exposure is calculated. The different options for exposure assessment are detailed in Section 5 of this document. In general, the method(s) used for estimating exposure and effluent effect should be comparable in terms of cost and required effort. Therefore, in the screening tier, as decisions are made on simple, inexpensive toxicity tests, simple exposure estimates such as assuming complete dilution at design low flow (e.g., 7Q10) should be employed. However, this assumption should be used cautiously since discharges often create shore-hugging plumes that mix slowly. As subsequent tiers of analysis are used, more sophisticated exposure analysis procedures, such as those described in Section 5, should be used.

A recently developed source of whole-effluent toxicity data can be useful in toxicity screening. The Complex Effluent Toxicity Information System (CETIS) is a data management system designed to catalogue the results of whole-effluent toxicity testing of municipal and

industrial wastewaters. CETIS is a data file within EPA's In-house Software (IHS) system. IHS was developed to augment the data management and analytical capabilities of STORET. A user's manual is available [11a, 11b]. Data are obtained from EPA Regional and State NPDES permitting programs and from published literature. CETIS can be used to assess a discharge in a specific industrial category for toxicity potential by comparing it to a similar discharge for which toxicity data are already available.

Whole-effluent Toxicity Screening

Several procedures can be used to screen for effluent toxicity. One option is to run acute toxicity tests, calculate the lowest LC50, divide it by available dilution, and compare it to the appropriate level of uncertainty. The ratio of LC50/IWC should be greater than this level of uncertainty. If it is not, the second tier of testing is required.

Another option for screening is to measure effluent toxicity using simple, fast screening tests conducted in the

Box 3-2
RECOMMENDATIONS FOR WHOLE-EFFLUENT TOXICITY SCREENING

- *Individual Dischargers* — Compare receiving water flow rate (in terms of whatever water quality-based design low flow is specified by the State) to average effluent flow rate.
  - —If dilution exceeds 10,000 to 1, and there is a reasonably rapid mix of the effluent outside of the rapid initial dilution area in the receiving water, then the effluent should be given a low priority for any further attention.
  - —If dilution is less than 10,000 to 1, or mixing is not rapid and toxicity within a plume is of concern, then toxicity screening tests should be performed.
  - —If dilution is between 1,000 to 1 and 10,000 to 1, or a poorly mixed effluent plume in a large receiving water (>10,000 to 1 dilution) is of concern, conduct acute toxicity screens as follows:
    1. Collect four to six effluent samples on one day (grab or short term composite), quarterly. Conduct screening tests (24-hour) in 100% effluent, using a daphnid and a fish, on each sample.
    2. If 50% mortality or greater is observed in three samples, the potential for toxicity is assumed and further testing is required.
    3. If 50% mortality or greater is observed for two or fewer samples, the discharge should be given a low priority for further analysis.
  - —If dilution is less than 1,000 to 1, conduct chronic toxicity screens (short term chronic tests are recommended) as follows:
    1. Collect four to six effluent samples (24-hour composite) on four to six successive days. Conduct static screening tests (seven-day) in 100% effluent, using a cladoceran and a fish, on each sample.
    2. If a 50% or greater effect is observed between controls and test organisms, the potential for toxicity is assumed and further testing is required.
    3. If less than 50% effect is observed, the discharge should be given a low priority for further analysis.

    Acute tests can be used in these dilution situations, but it should be noted that there will be cases where no acute toxicity is measured but the effluent is chronically toxic.
  - —Where dilution is less than 100 to 1, the use of a toxicity-testing-based screening procedure is not recommended. Screening has already been accomplished through dilution analysis. Even in discharge situations where no toxicity is observed in screening tests, the narrow margin between effect concentration and available dilution suggests more complete effluent toxicity characterization is mandatory. If uncertainty factors are applied in a 100 to 1 discharge situation, dilution alone would mandate further testing. Where very limited dilution is available, it is recommended that toxicity-testing screening be skipped and the discharger be required to begin DEFINITIVE DATA GENERATION procedures. An example of this situation is described in Section 8.
- ***Ambient Toxicity Analysis*** — **Use ambient toxicity analysis to identify areas of instream toxicity associated with specific dischargers. This analysis may be most useful when conducted by the regulatory agency, but dischargers may be required to conduct the tests in conjunction with effluent tests. A systematic plan for identifying problem areas is recommended. This procedure is useful for multiple source discharge situations. The analysis should be conducted concurrently with discharge-specific screening and must be done at low flow conditions. A procedure is described in Appendix C.**

absence of serial dilutions at 100% effluent concentration. If a pre-specified toxicity level is observed, the potential for toxic impact exists and further analysis is conducted. Uncertainty factors are not used in this option because LC50s are not generated. Also, dilution factors (described in Box 3-2) serve the same function as the uncertainty factors used with the first option described.

One recently developed procedure, ambient toxicity testing, is useful in screening stream reaches and discrete watershed areas for existing instream toxic conditions. The procedure, described in Appendix C, uses short term chronic toxicity tests to measure the toxicity of samples of receiving water taken above, at, and below (or around) outfalls. The utility of this approach in screening is that actual instream toxicity is measured, signifying the probability of toxic impact. No extrapolation for exposure or ACR is needed. Further, impact from multiple-source discharge situations, which may not be apparent from individual outfall screening data, is identified. Finally, the technique can provide an assessment of the persistence of effluent toxicity. The procedure must be conducted during an appropriate, design low-flow period.

Chemical-specific Screening

There are two levels of analysis for chemical data. Both are based on the margin between measured chemical concentration and the State water quality standards or the national water quality criteria. The first is simple dilution analysis and comparison with State standards or national criteria. Where high background concentrations exist, the concentration in the receiving water prior to discharge should be included in the calculation.

The second level of analysis is to use fate models to estimate persistence. The EPA has recently supported the development of a chemical-specific screening approach that is applied to lakes, rivers, estuaries, and coastal systems [12]. It contains methods for estimating waste loadings from urban and rural point and nonpoint sources and for predicting the levels of contaminants in waterbodies. Methods are included for conventional pollutants (nutrients, dissolved oxygen, biochemical oxygen demand, coliforms, temperature, sediment transport, and salinity), metals (currently being expanded), and the organic priority pollutants. Considerable supporting information is provided in the manual, particularly on rate constant information for toxicants. The tools are intended to be used with a desk top calculator or microcomputer.

EPA's recommendations for chemical-specific screening are presented in Box 3-3.

Box 3-3
RECOMMENDATIONS FOR CHEMICAL-SPECIFIC SCREENING

- Examine existing data to determine presence of specific toxicants for which criteria, standards or other toxicity data are available. Sources of data include:
  - —Permit application forms, DMRs, and permit files.
  - —Pretreatment industrial surveys.
  - —STORET for U.S. Geological Survey (USGS) flow data and ambient monitoring data.
  - —Industrial effluent guidelines development documents.
  - —The Treatability Manual [13].
- Use simple dilution assumptions to compare effluent data and corresponding estimated instream concentrations to the concentrations specified in the national water quality criteria, State water quality standards, or the literature for the toxicant(s) of concern.
- Use simple fate models and screening procedures [12] for situations where better estimates of instream concentrations are needed.
- Follow the procedures for screening for human health problems described in Section 4.
- In making screening decisions from limited data, an uncertainty factor of 10 or 100 can be used to account for a toxicant's potential concentration variability. In this way, effluent variability is addressed in the chemical-specific screening process.

Tier 2. Definitive Data Generation

Whole-effluent Definitive Data Generation

Once screening has indicated the potential for toxic impact, further data may be needed to determine the degree of impact caused by the discharge. In this tier, whole-effluent toxicity testing is one of two procedures that may be conducted to provide the data necessary to measure the worst case ambient toxicity conditions or to estimate those conditions through a limited data extrapolation process. Identifying and quantifying the worst case conditions require an analysis of toxic effect and exposure assessment (detailed in Section 5) and an analysis of the variability associated with these conditions at the site.

EPA's recommendations for generating whole-effluent definitive data are presented in Box 3-4.

Chemical-specific Definitive Data Generation

The recommended procedures for monitoring heavy metals and organic chemicals are described in this part of the Effluent Characterization Section. Analyses for all toxicants are described in Standard Methods of Water and Wastewater Analyses (ASTM, 15th edition, 1982). Detailed recommendations on the design of appropriate sampling programs for rivers, lakes, and estuaries are included in the Monitoring and Data Support Division's Technical Guidance Manual for Perform-

Box 3-4
RECOMMENDATIONS FOR WHOLE-EFFLUENT DEFINITIVE DATA GENERATION

Requirements for toxicity analysis are site specific. Recommendations for testing in this section will be presented in terms of eliminating levels of uncertainty. Analytical procedures will be presented together with those uncertainty factors which can be eliminated if those procedures are used. Again, a simple relationship can be applied to determine whether to require more data, whether to stop testing and begin the process of establishing permit conditions, or whether to cease analysis because of a wide margin between toxicity and the IWC:

$$\frac{\text{LC50 or NOEL (in \%)}}{\text{IWC}} > \text{level of uncertainty}$$

where the level of uncertainty equals the uncertainty factors multiplied together. It should be stressed that this equation is used to evaluate the need for additional analysis. The relationship is not the basis for the development of permit limits or conditions. These procedures are discussed in Section 6.

- *Initial or "Baseline" Testing*
  - *Tests* — Conduct acute toxicity tests on two species. A fish and an invertebrate are recommended.
  - *Frequency* — Conduct monthly on grab or composite samples, whichever is appropriate. Where court orders, administrative orders, or legal deadlines require permits to be issued within a shorter timeframe, schedule these tests to conform with the deadline.
  - *Uncertainty factors* — 10X to 100X for effluent variability, 10X for species sensitivity variability, and 10X for ACR where chronic toxicity is of concern, but no chronic data are available. *Level of uncertainty = 1,000 to 10,000.*
- *Eliminate Effluent Variability Factor*
  - *Tests* — Conduct acute toxicity tests on two species. A fish and an invertebrate are recommended.
  - *Frequency* — Examine manufacturing processes, treatment plant design and retention times (actual, not design), and variability of measured parameters to estimate the variability of toxicity. It may be impossible in a practical sense to assess toxicity variability, but an estimate should be made initially. This factor is perhaps the most important source of uncertainty in the toxicity assessment process.

    For effluents exhibiting short term (12 to 48 hours) variability, measure the highest toxicity during a short period of duration (one discharge cycle or a 24-hour period). Four to six samples (grab or short term composite) taken during this period should be tested. Tests are conducted monthly for at least one year. Another option is to require a continuously pumped, flow-through acute test on each species. This will integrate the effects of a variable toxic concentration but will not provide a quantification of effluent variability (see Appendix B).

    For effluents exhibiting suspected long term variability, schedule testing frequency to conform to expected changes (weekly, monthly, seasonal, process changes) in effluent composition, if known (see Appendix B). For long term variability, a year-long monitoring program may be required to determine long term variability.
  - *Uncertainty Factors* — 10X for species sensitivity and 10X for ACR where chronic toxicity is of concern but no chronic data are available. *Level of uncertainty = 100.*
- *Eliminate Species Sensitivity Factor*
  - *Tests* — Conduct acute toxicity tests on a total of three to five species. The species should be representative of several different groups including fish, invertebrates, and plants.
  - *Frequency* — Same as above.
  - *Uncertainty Factor* — 10X where chronic toxicity is of concern but no chronic data are available. *Level of uncertainty = 10.*
- *Eliminate ACR Factor*
  - *Tests* — Conduct short term chronic toxicity tests on three species. Since the available test procedures short enough to be practical in effluent characterization are limited, to eliminate the species sensitivity factor where chronic testing is mandatory, the use of *Ceriodaphnia*, fathead minnow growth test, or other short term chronic tests currently available is recommended.
  - *Frequency* — Same as above.
  - *Uncertainty Factor* — Level of uncertainty equals one if these data are generated.
- *Other Considerations*
  - *Use of Dye Studies* — Dye studies are strongly recommended for effluent characterization. They are relatively inexpensive and provide data needed to plan an assessment of an effluent's impact on the receiving water. A dye study should be included in any Tier 2 analysis unless mixing is known to be rapid and complete. Procedures for conducting a dye study in an effluent characterization are described in the Lima, Ohio, report [5].

ing Wasteload Allocations [13-15]. In general, plug-flow sampling is recommended for rivers, and vertical profile monitoring is recommended for lakes. A combination of slack water and intensive surveys is recommended for lakes, rivers, reservoirs, and estuaries.

It is important for each toxicant, whether it is a heavy metal or an organic chemical, that the analytical equipment achieve the best resolution and lowest detection limit possible. As a minimum, the detection limit should be at least as low as the concentration of the water quality standard or criterion. EPA generally recommends high-resolution, capillary-column gas chromatographs (GC) for organic chemical analysis and graphite-furnace atomic adsorption (AA) for metals.

If heavy metals are being monitored for fate and transport modeling, bed sediment should be analyzed for the following parameters: solids concentration, particle size distribution, total toxicant concentration, dissolved toxicant concentration, and percent water in sediment. If organic chemicals are being investigated, an additional analyisis must be conducted that measures the weight fraction of organic carbon in particles less than or equal to 50 micrometers (μm) (fine) and in particles greater than 50 μm (coarse). This organic carbon measurement should be determined using a wet oxidation method if high carbonate soils are being analyzed, and a gravimetric analysis if low carbonate soils are present. These analyses are described in soil chemistry textbooks.

If heavy metals are being monitored for fate and transport modeling, the water column should be analyzed for the following parameters: total toxicant concentration, dissolved toxicant concentration, total suspended solids (TSS), total organic carbon (TOC) and dissolved organic carbon (DOC), pH, temperature, hardness and total alkalinity, dissolved oxygen (DO), ammonia, chlorine residual, and specific conductivity. If organic chemicals are being monitored and modeled, additional analyses should be conducted: weight fraction of organic content in suspended particles less than or equal to 50 μm and in suspended particles greater than 50 μm and secchi disk depth. The TOC and DOC data indicate humic acid content of waters and the potential for metal complexing. The pH, temperature, hardness, and total alkalinity data are needed to calculate the appropriate water quality criteria for metals and organic chemicals. The DO, ammonia, and chlorine residual measurements can be used to interpret observed toxicity and biotic status. Conductivity measurements confirm model representation of transport, and secchi disk depths indicate the potential for photolysis of organics.

Monitoring of toxic substances requires special quality control procedures beyond those necessary for monitoring conventional parameters because toxicants generally occur in trace concentrations and are frequently unstable. The most up-to-date sampling and handling procedures recommended by EPA for a number of toxic and conventional parameters are detailed in a recent publication [17].

Samples that will be filtered for particulate and dissolved fractions should be delivered to the laboratory for filtration within one or two hours maximum for metals and filtered and preserved on site. For organic chemicals that may photolyze and biodegrade readily, samples should be exposed to the atmosphere as little as possible to avoid the loss of volatile compounds. Sample bottles must be clean and made of materials that will not contaminate the samples, either plastic or glass, depending on the analyses to be performed. Replicate samples should be taken and analyzed to assess the variability of measurements caused by sampling techniques and site heterogeneity. A blank and one sample spiked with each of the toxicants should be analyzed with each group of samples.

The documentation of all field and laboratory procedures is needed to ensure the defensibility of toxics monitoring data. A sample observation sheet should be filled out for each station providing observations on surface conditions. For each sampling location, a sample log book should be kept with a record of the *in situ* results and the numbers and times of water column measurements. All samples should be properly labeled and numbered with preprinted forms and labels. Additionally, a log book should be kept in the laboratory in order to record each sample as it arrives in the lab and to document the analytical results.

Once these data have been generated (the responsibility of the permittee), the discharge is analyzed for water quality problems using the procedures discussed in Section 5.

EPA's recommendations for generating chemical-specific definitive data are presented in Box 3-5.

Use of Toxicity Testing in Multiple-source Discharge Situations

No formal definition exists to determine what a multiple-source discharge situation is. One definition could be that it exists where impact zones overlap.

Where more than one discharge is contributing to toxic impact, additional data may be needed. Testing that provides the regulatory authority with the information necessary to separate the relative impact of each source can be conducted. In multiple-source discharge situations, additivity, antagonism, and persistence of toxicity may become more important. Therefore, more detailed testing designed to measure these factors would be warranted.

Two options for assessment can be identified in these cases. The regulatory agency may decide to handle each source separately. The procedures described

Box 3-5
RECOMMENDATIONS FOR CHEMICAL-SPECIFIC DEFINITIVE DATA GENERATION

The choice of chemical-specific data generation procedures is based on the type of modeling analysis found to be most appropriate for the specific discharge situation. Steady state analysis is more stringent since it assesses the worst case exposure situation (where a proper, toxicology-based, critical low flow is used) and requires comparable treatment. Dynamic or probabilistic modeling will generally result in more cost-effective treatment requirements because these procedures more accurately assess actual exposure conditions. A complete discussion of the choice of modeling techniques is found in Section 5.

- ***Limits to be Based on Steady State WLAs*** **— Conduct critical condition sampling as appropriate for rivers, lakes, and estuaries. GC analysis (for organics) and AA analysis (for metals) for the specific toxicant(s) of concern should be conducted at some designated critical period.**
- ***Limits to be Based on Dynamic or Probabilistic WLAs*** **— Conduct comprehensive sampling of effluent. Multiple grab or composite effluent samples should be taken over a critical season or the entire year to define effluent variability. GC analysis (for organics) and AA analysis (for metals) for the specific toxicant(s) of concern should be conducted using the recommended frequency.**

previously would be used. Another option is to treat each discharge as an interactive component of the whole system. If this option is selected the regulatory agency could follow the testing procedures presented in Box 3-6. Again, testing is the resposibility of the permittees. It is assumed that screening has been conducted, and the potential for toxic impact has already been shown.

Use of Effluent Characterization Data and Decision Criteria

Where levels of uncertainty for the last testing tier completed are not exceeded, the discharger should be given a low priority for regulatory actions designed to control toxicity. Where the level of uncertainty is exceeded by the toxic level measured, the data generated should be used as the basis for toxics control. The process of exposure assessment and permit issuance is described in Sections 5 and 6.

Bioaccumulation

Bioaccumulation is the accumulation of chemicals in the body of an animal to concentrations greater than the ambient concentration. Bioaccumulation occurs both through uptake across the gill membranes (bioconcentration) and through ingestion of contaminated food (biomagnification). Some chemicals can accumulate to hazardous levels in fish when the concentration in the water is below a toxic level. Therefore, effluents that have an acceptable level of toxicity may contain hazardous concentrations of bioaccumulative pollutants.

The potential impacts of bioaccumulative pollutants include: 1) accumulation of toxic pollutants in aquatic biota resulting in effects on survival or reproduction, 2) economic impacts resulting from fish tainting, and 3) exposure of humans to hazardous chemicals through drinking water or consumption of contaminated fish or shellfish. Some chemicals bioaccumulate but have no apparent adverse effects. However, any chemical that has high potential for persistence and bioaccumulation should be a matter of concern until it can be demonstrated that there are no adverse environmental and human health effects resulting from discharge.

Chemicals that bioaccumulate include metals, organic compounds, and organometallic compounds. Metals and organometallic compounds are adequately addressed through the EPA ambient criteria for human health and aquatic life. Similarly, application of the EPA ambient criteria for organic compounds will adequately control those compounds because the criteria are based on bioaccumulation where appropriate. The problem in addressing effluent discharges involves organic compounds that are not among the priority pollutants for which there are criteria.

The first step in addressing bioaccumulative pollutants in effluents is to determine whether or not such pollutants are present. The second step determines if such pollutants are hazardous. The final step calculates an acceptable discharge rate. Since bioaccumulation is perhaps the most important human exposure route, bioaccumulation assessment is discussed in Section 4, Human Health.

Box 3-6

RECOMMENDED MULTIPLE-SOURCE TOXICITY TESTING PROCEDURES

- *Tests*
  - —For effluent-dominated (a suggested definition is where the effluent(s) make up 1% or greater of the flow) receiving waters (assuming total flow input from all sources including other effluent flows is measured), conduct chronic toxicity tests following the testing procedures previously described.
  - —For stream-dominated (where the effluent(s) make up less than 1% of the flow) receiving waters (assuming total flow input from all sources including other effluent flows is measured), conduct acute toxicity tests following the testing procedures previously described.
  - —An additional data requirement in multiple-source discharge situations is the assessment of *relative* and *absolute* toxicity of each source so that appropriate permit conditions can be set for individual dischargers. The following procedure is suggested.

    1. Conduct one set of toxicity tests on the effluents using a control dilution water, either from an upstream, uncontaminated receiving water station of known high quality with a similar chemical makeup or using reconstituted water. This gives an *absolute* toxicity measurement of the effluent.
    2. Run a parallel set of toxicity tests on the effluent using a dilution water taken directly upstream from the point of discharge. This dilution water may be contaminated with upstream effluents or other toxicant sources. The purpose of this test is to establish toxicity measurements of the effluent as it is mixed at its point of discharge. This is a *relative* effluent toxicity measurement. The relative testing procedure could result in a change in the standard concentration-effect curve generated by the testing.

       The dilution water for the relative toxicity test may cause significant mortality (or growth or reproductive effect) at the lower effluent concentrations (including the 100% diluent control concentration) if the diluent from the receiving water is toxic. Of course, it will be toxic if an upstream source is toxic. This mortality does not invalidate the test. Instead, analysis of toxicity trends resulting from the relative toxicity tests can be used to assess the effluent's toxicity in relation to other sources and ambient receiving water conditions. However, there must always be a control with no toxicity in the dilution water for quality assurance and determination of absolute toxicity of the effluent.
    3. Conduct ambient toxicity tests to: 1) determine whether or not the effluent has a measurable toxicity after mixing, 2) measure instream persistence of toxicity from all sources contributing to instream toxicity, and 3) determine combined instream toxicity resulting from the mixing of multiple, point and non-point sources of toxicity. See Appendix C for a discussion of ambient toxicity testing procedures.

       The ambient testing can be conducted by the regulatory agency or can be required of each discharger who would then be responsible for the upstream and downstream areas around their outfall. It should be conducted during a low flow period.
- *Frequency*
  - —Conduct tests at appropriate frequencies as described in the previous section. All testing should be conducted simultaneously by each discharger, if possible. At a minimum the tests should be conducted concurrently starting within a short time period (one to two days). Repeated ambient toxicity analyses will be desirable when variable effluents are involved. Effluent toxicity data showing variability can be used to assess what frequency will be most applicable. The level of repetition for variability analysis should be similar to that used in effluent variability analysis.
- *Other Considerations*
  - —Dye studies of effluent dispersion for rivers, lakes, reservoirs, and estuaries are strongly recommended. This allows analysis of effluent concentration at the selected sampling stations above and below the discharge points.
  - —The procedures suggested in this multiple-source section are based on actual multiple-source site investigations conducted under the Complex Effluent Toxicity Testing Program. Site reports from that study can be used to obtain further description of the toxicity testing procedures used to analyze multiple-source toxic impact [5,6].

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4. Human Health Hazard Assessment

Overview

The assessment of human health hazard, as with the assessment of aquatic impact, depends on two factors: 1) the toxicological properties of the pollutants discharged and 2) the degree of exposure of the target populations to the pollutants. However, health hazards are considerably more difficult to assess than aquatic impacts. The reasons are that latent effects can occur many years after exposure, toxicological data are always indirect because of the inability to test humans, and routes of exposure are difficult to project, particularly since there is almost always a treatment system between released effluent and human exposure. A variety of factors other than those mentioned above can also influence toxicity.

Assessing effluent-related human exposure is a relatively new area for regulatory agencies. Most of the effort has been concentrated on drinking water, the industrial workplace, and other routes of exposure such as food. Most of these problems have been addressed using pollutant-specific chemical analysis, toxicological information, and exposure analysis to assess and regulate exposure. This approach tends to break down for effluents because of the diversity of chemicals present and the lack of analytical methods, toxicological information, and means to assess interactions. The use of biological tests to test effluents for health effects is an attractive alternative; however, the degree to which such tests reflect potential human health hazards is not known. Although biological testing using short term assays cannot yet be used to quantify health hazards, they can be used for screening to identify potential hazards.

This section will provide background information on health effects, a review of chemical vs. biological approaches, recommendations for screening and priority setting, and recommendations for assessment and control.

Background

Various toxicological effects in the human population are thought to be caused by environmental exposure to chemical substances. Toxic effects will be classified into two broad categories: 1) genotoxic effects such as carcinogenicity and heritable mutation and 2) systemic or target organ effects that impair the functioning of specific organ systems.

Genotoxicity is the ability of a substance to damage an organism's genetic material (DNA). Damage to the DNA of reproductive cells can impair reproductive ability or can produce a change in the DNA structure that could be passed on to offspring as a heritable mutation. A chemical must interact with germ cells and permanently change the linear structure of the DNA and pass that change to offspring before it can be classified as a human mutagen.

Damage to the DNA of non-reproductive cells can result in uncontrolled cell proliferation leading to cancer. Many classes of carcinogens are electrophilic or metabolized *in vivo* to form electrophilic reactants that may ultimately interact with cellular DNA. This is thought to be the initiating and critical step for genotoxic carcinogens. Other subsequent changes are still necessary before a malignancy appears. The fact that many carcinogens are also mutagens has led to the idea that mutation tests may be predictive of carcinogenic properties. This generalization on the action of carcinogens, however, has not been demonstrated to apply to all chemical carcinogens.

Systemic toxicity is the ability of a substance to affect the function of one or more organ systems. Although a comprehensive discussion of these effects is beyond the scope of this manual, the following target organ toxicities have been observed from oral exposure to chemical substances (the target organ is indicated in parentheses): hepatotoxicity (liver), renal toxicity (kidney), cardiovascular toxicity (heart and blood vessels), neurotoxicity (brain and peripheral nerves), reproductive toxicity (testes or ovaries) and hematopoietic effects (bone marrow). A subchronic or chronic whole animal testing protocol is typically employed to detect target organ toxicities among the major organ systems. Toxicological results from mammalian studies have been generally accepted for purposes of directly estimating potential hazards to man.

Risk assessment must include an estimate of exposure. Ingestion of drinking water is the most obvious route of exposure to waterborne pollutants since a broad segment of the population could be affected. There are other less obvious but potentially significant routes of exposure. Fish and seafood contaminated through the bioaccumulation of pollutants by aquatic organisms may be a significant source of exposure. Absorption through the skin that may occur during bathing or swimming, and inhalation exposure that may occur during breathing of humidified air in the home are also pathways for toxic substances.

Bioaccumulation

Bioaccumulation is a potentially significant route of exposure to human populations. Pollutants that are health hazards may also cause severe impacts on aquatic organisms and other wildlife through bioaccumulation.

Of the two processes that constitute bioaccumulation (bioconcentration and biomagnification) bioconcentration or uptake through the gills is the easier process to study. Therefore, most evaluation techniques are based on bioconcentration with the inference that, by limiting bioconcentration, biomagnification is also controlled. The measure of a chemical's tendency to bioconcentrate is termed the bioconcentration factor or BCF and is defined as:

$$\text{BCF} = \frac{\text{accumulated concentration in tissue}}{\text{exposure concentration in water}}$$

Bioconcentration has been extensively studied and the following conclusions have been drawn: 1) the BCF for a compound is constant over a wide range of exposure concentrations, 2) compounds with low BCFs reach steady state residue concentrations quickly, whereas compounds with high BCFs may never reach steady state, and 3) the BCF is correlated very well with the n-octanol/water partition coefficient. This correlation allows the n-octanol/water coefficient to be used to estimate the BCF of a compound (see Figure 4-1). The partition coefficient also provides a way to separate potentially bioaccumulative pollutants from other pollutants.

Status of Chemical vs. Biological Approaches

Most health-related toxicological information has come from the testing of biological organisms. Almost all of this work, however, has been done on single chemicals and not on mixtures. This is because of the experimental simplicity and the need to assess risks of single chemicals under the Toxic Substances Control Act (TSCA), Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), and Food and Drug Administration (FDA) statutes. Results from toxicological experiments on a single chemical are combined into a dose response curve for that chemical describing its toxicological potency. This is then combined with an exposure model to estimate risk.

A body of toxicological knowledge has been built to the point where an untested chemical's toxicological properties can be estimated by comparing its structure to the structure of tested chemicals. This field of science is called structure activity relationship (SAR) and is rapidly developing. There are computer programs that can estimate toxicological properties. However, the use of SAR requires considerable expertise and is not recommended as a casual exercise. It has also come under some criticism and is not fully accepted, especially as a regulatory basis for toxics control.

Effluents present a difficult problem, in contrast to single chemicals, because they may be composed of a mixture of chemicals. A second problem is that the constituents of the effluent may interact to change the effects of a chemical compared to its effects in "pure" form. These problems severely limit the approach of identifying the individual chemicals in the effluent and conducting a risk assessment for each chemical independently. A logical alternative is to test an effluent directly using biological tests. This approach also has severe limitations. First, very little research has been conducted on complex mixtures (of which effluents are a good example) to determine how various tests respond. Second, the higher animal testing that is needed to quantify the dose-response relationship (the "potency" of the effluent) would be extremely costly. Third, effluent testing presents several practical problems such as the need to handle the continual change in composition typical for most effluents, the need to concentrate samples to get a dose-response curve, and the need to deal with interferences from cytotoxic components of the effluent.

Some of the biological tests used to assess the toxicology of pure chemicals can be used practically to test effluent samples. These are the short term tests used in the first tiers of the single chemical evaluation process. In that process, the results of these tests are used to decide whether more definitive (and much more expensive) testing is needed. Therefore, they constitute screening tests. These short term tests are based on cellular level response and do not measure the actual response that higher organisms, such as mice or humans, would exhibit to the toxicant. However, they do indicate whether the test substances are biologically active and provide some measure of that activity. Each test system measures a different type of response so that a battery of several tests is usually considered necessary. Research is currently underway within EPA to evaluate the use of various biological tests for effluent testing.

The chemical-specific approach should be used to quantify risks accurately and to determine treatment requirements in the assessment of effluent health hazards. It is important to note that the priority pollutant list is not sufficient for chemical-specific analysis because it is so limited in scope. There are many more hazardous toxicants discharged than are listed. Identifying pollutants requires techniques such as process evaluation and extensive analysis of GC/MS data. It can be extremely expensive, however, to identify important chemicals or classes of chemicals and determine their toxicological properties. Therefore, a priority setting process is needed. For setting priorities, biological tests

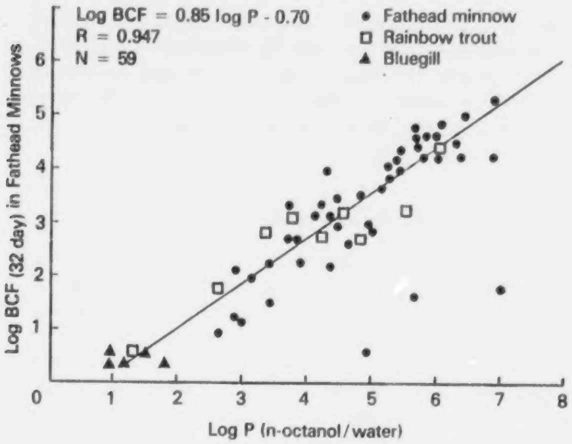

Figure 4-1. Relationship between the bioconcentration factor of 54 organic chemicals in fish to the log P of the chemical.

along with a chemical review of expected effluent components can provide the most cost-effective approach.

Priority Setting

The evaluation of a discharge for potential hazard consists of two components — estimating exposure and estimating the toxic effects of the effluent. The following example of a priority setting approach is intended to provide a starting point for more detailed State-specific priority ranking systems.

This example approach consists of evaluating each component (exposure and effects) and assigning a value of high, medium, or low depending on the magnitude of potential risk. This provides a two-way matrix for ranking sources. The matrix could then be used to rank sources in order to assign priority for further investigation. Alternatively, decision criteria could be developed in advance so that, as soon as a source is screened, an appropriate action could be taken.

Exposure

Three major routes should be assessed when evaluating potential human exposure to a toxicant in water. These are ingestion by drinking water, ingestion of contaminated aquatic organisms, and intake through contact recreation. Exposure due to drinking water requires knowledge of the concentration of pollutant and an estimate of the quantity of water drunk. This latter value is generally assumed to be two liters per adult or one liter for a child. If the concentration is to be based on dilution calculations rather than direct measurement, it should be done using a low flow estimate such as the 30-day average low flow with a five-year return period.

Consumption of aquatic organisms can be estimated using the national average consumption rate of 6.5 grams (g)/day. If the local population is known to consume more or less fish than this, the exposure assessment would be more accurate if consumption was based on local fish sales or harvest.

Finally, the location and extent of use of recreation areas where exposure might take place should be documented. Types of intake through this route might include dermal absorption, accidental swallowing, and cuts. Generally, only crude estimates can be made of this route.

Effects

Two types of information should be collected in order to evaluate the effects that pollutants contained in the discharge may have. First, the discharger should provide an estimate of all chemicals that may be present in the wastewater. This can be accomplished by analyzing the manufacturing processes and listing all chemicals used, produced, or created as byproducts. Where the effluent is complex, an identification of classes of compounds thought to be present would suffice for this screening analysis. Some attempt to estimate the concentration or volume of each chemical should be made. This information can then be reviewed against the toxicology data bases to identify which chemicals are potentially hazardous. When only a class of compounds is identified as present, the toxicology information available for any members of the class could be used to evaluate that component of the effluent.

The discharger should also, in addition to conducting a chemical review, be required to conduct short term biological tests. Biological test results are an important supplement to a chemical review because the vast majority of chemicals lack toxicology information and because of the inability to identify many of the chemicals in complex wastewaters. A variety of toxicity tests exist and are accepted for assessing potential human health hazards [1,2], but most of these have not been used for effluent toxicity impact assessment because they are either impractical (too expensive or time consuming), or not sufficiently well developed to provide reliable or meaningful data. Recently, a variety of short-term tests that provide relatively rapid results at affordable costs have become sufficiently well developed to be considered for use in screening health hazards associated with complex effluents [3,4].

A battery of tests is needed in order to test for the major types of potential impact. Many of the genotoxic effects can be tested for using bacterial or cell culture techniques. Systemic or target organ effects, however, can only be tested using whole animal studies. The best means to assess systemic effects is by using subchronic toxicity procedures designed to determine the effects that may occur with repeated exposure over a part of the average life span of an experimental animal. However, such studies are expensive ($100,000 and over) and probably beyond the cost constraints for a screening level analysis. These methods are discussed later in the section on risk assessment. There are no inexpensive short term tests that can assess the potential for systemic toxicity effects.

Several short term assays have been developed to assess genotoxic effects. The two major types of effects are point mutations and clastogenic activity (chromosomal breakage). To assess point mutation effects, the Ames test is recommended. The Ames test can be followed by a mammalian gene mutation test using cells in culture if confirmation is desired. To assess clastogenic effects either the mammalian sister chromatid exchange test or a mammalian cell chromosomal aberrations test should be conducted. Both of these tests typically use Chinese hamster ovary cell cultures and involve cytologic examination after exposure to determine if chromosomal effects are evident. The Organization of Economic Cooperation Development (OECD) test methodology is recommended [5]. EPA's Office of Toxic Substances and Office of Pesticides Programs also have published test methods (see References [17] and [18], Section 1) that are consistent with the OECD tests.

Some effluent samples may not need to be concentrated to measure a response, but the majority of samples should be concentrated. When samples are concentrated, the response should be calculated in terms of the preconcentrated sample (e.g., Ames results would be reported as revertants per milliliter (ml) of original sample). Caution should be taken when collecting waste effluent, concentrating complex mixtures, and extracting the chemicals for toxicological testing. Ion exchange and liquid/liquid extraction techniques are most commonly used and are recommended. If an effluent is extremely variable, screening assays should be repeated periodically to ensure that potentialy hazardous discharges are detected.

Test results should be evaluated to decide whether a high, medium, or low toxic effect potential exists. Considerable scientific judgment is required. For example, Ames results are expressed as the number of revertants per ml original sample. A response twice the solvent blank (control) is considered a positive. A scale could be constructed where less than twice the control is negative, up to 100 revertants/ml is a "low," between 100 and 1000 revertants/ml is a "medium," and greater than 1000 revertants/ml is a "high" response.

Evaluation

The State or Region should review the results of a screening program and establish actions for each level of potential risk indicated. For example, a discharger with either a high exposure risk or a high effects risk might automatically be required to conduct a detailed assessment or institute controls. A medium risk in both exposure and effects might require further review of the data and a case-specific decision about whether to require additional assessment. A medium and a low risk might indicate the need for limited testing to ensure that the low is really indicative of the risk. Low risk in both exposure and effects should receive low priority for further assessment.

Assessment and Control

When screening has indicated a high potential for health hazard, further assessment is required. A chemical-specific approach is recommended to evaluate the discharge constituents. The first half of this process involves characterizing the composition of the wastewater. It should be recognized from the outset that it is usually impossible to identify all the compounds contained in a wastewater. Typically, only a small fraction of the total organic carbon (TOC) can be accounted for as specific chemicals. Therefore, a substantial effort should be put into identifying constituents through means other than chemical analysis. The best way to accomplish this is through a detailed process evaluation.

A process evaluation is a study in which components in the wastewater are determined from an analysis of feedstocks, manufacturing processes, products, byproducts, and pollution control in place. The result is a list of compounds or classes of compounds with a high probability of being present in the wastewater. Chemical analysis should also be conducted for not only the priority pollutants but also nonpriority pollutant peaks.

Particular attention should be paid to bioaccumulative pollutants. There are three approaches to identifying non-priority bioaccumulative pollutants in effluents. Aquatic biota can be collected upstream and downstream of an effluent outfall and analyzed for specific compounds or any compound present in downstream biota that is not present in upstream biota. This approach is expensive. There is also the problem of locating sources in multiple-source situations. However, this approach is the only way to assess bioaccumulation.

Another approach is to rely on the list of compounds used or manufactured that may be present in the effluent. For each compound, the n-octanol/water partition coefficient can be looked up or estimated from chemical SARs. EPA recommends that any compound for which the logarithm of the partition coefficient (log P) is greater than 3.5 be flagged for further evaluation and possible control [6]. Computing the log P of industrial chemicals from structure has been the objective of a six-year project at Pomona College. A computer program that accurately calculates log P is available to all Regions and States. Contact Dr. Gilman Veith in EPA's Duluth Laboratory, 6201 Congdon Blvd., Duluth, MN 55804, (218) 727-6692 or FTS 783-9550.

Third, if the effluent is too complex to list all compounds, the effluent can be analyzed using high pressure liquid chromatography (HPLC). The partition coefficient of a compound is directly related to its retention time using HPLC separation systems [7]. Since water is the carrier solvent, effluent samples may be directly injected into the HPLC system, and those compounds corresponding to log P greater than 3.5 can be detected or collected and identified [8].

To determine if bioaccumulative compounds are present, the HPLC can be connected to an infrared detector to measure peaks. To identify compounds, the peaks can be collected and subsequently analyzed using GC/MS methods.

Once pollutants or classes of chemicals present in the wastewater have been identified, the next step is to evaluate their toxicological properties. Sources of information that may be useful are listed below.

- Water quality criteria — Developed by the Office of Research and Development (ORD) and available through EPA Office of Water Regulations and Standards, (202) 245-3030. Criteria documents are available for 65 compounds or classes of compounds which review toxicology and health effects and, if sufficient data exist, present ambient criteria to protect human health.
- The Carcinogen Assessment Group (CAG) List — The CAG of EPA ORD has published a draft list of chemicals that are known to be potent carcinogens. A revised list with potency factor and hazard ranking is currently under development. Contact CAG (RD-689), Washington, D.C., (202) 382-7341.
- Acceptable daily intakes (ADIs) for non-genotoxic pollutants — The Environmental Criteria and Assessment Office of EPA ORD is developing two publications. One is a methodology and ranking of 44 chemicals based on chronic toxicity. The other is a summary of ADIs for oral exposure. They are currently in draft. Contact the Office listed above at Mail Stop MD-52, Research Triangle Park, North Carolina 27711, (919) 541-4173.
- Health Advisories — These are issued by both the EPA Office of Drinking Water and the National Academy of Science. The EPA Office of Drinking Water also establishes National Interim Primary and Secondary Drinking Water Standards (see 40 CFR 141). Contact Dr. Khris Khana, Office of Drinking Water, Washington, D.C., (202) 382-7588.
- Chemical Information System (CIS) — This extensive data system is a collection of several data bases. The user is able to search all data bases using the structure, substructure, and full or partial name of over 250,000 substances. Formerly supported by EPA, it is now operated by Chemical Information Services, Inc., Dr. Al Fein, 7215 York Road, Baltimore, Maryland 21212, (800) 247-8737 (Hotline).
- National Library of Medicine (NLM) — The NLM maintains two data bases of toxicology information: Toxicology Information Online (TOXLINE) and Registry of Toxic Effects of Chemical Substances (RTECS). These data bases are available through the CIS system or can be used separately by contacting Medlars Customer Service, 5600 Rockville Pike, Bethesda, Maryland 20209, (301) 496-6193 or 638-8480.
- Dialog — This data system is designed primarily to locate and retrieve bibliographic information and abstracts of articles. Contact Dialog Customer Service, 3460 Hillview Avenue, Palo Alto, California 94304.
- QSAR — This is a computer program that will predict parameters useful for exposure assessment such as BCF, biodegradation and hydrolysis rates, etc. and will screen chemical structures for features that may contribute to the induction of cancer. Contact Lee Faulkner, Director, Center for Data Systems and Analysis, Montana State University, Bozeman, Montana 59717, (406) 994-4481.
- Michigan's Critical Materials Register — Chemicals are numerically scored as to their hazard, and those posing high hazard are included in the register. Contact MDNR Office of Toxic Material Control, PO Box 30028, Lansing, Michigan 48909, (517) 374-9640.
- Office of Pesticides Programs estimates of ADIs for various pesticides.
- WHO/IPCS Environmental Health Criteria Documents.
- WHO/IPCS Monographs on Pesticides.
- FDA ADIs.

Toxicology information should be used to narrow the field of pollutants to be investigated. Chemicals and classes of chemicals on this shortened list are then evaluated in terms of the potential exposure to target populations. Exposure assessment is the "determination or estimation . . . of the magnitude, frequency, duration, and route of exposure" (Proposed Guidelines for Exposure Assessment; 49 FR 46304, November 23, 1984). Two principal routes of exposure should be investigated for each chemical or class of chemicals — drinking water and bioaccumulation. Each of these routes is discussed below.

The means for evaluating the drinking water route of exposure depends on how the toxicological information is expressed. If there is an acceptable drinking water concentration for the chemical, the object is then to estimate the concentration at drinking water intakes. Two factors should be considered: dilution and degradation. Concentration after degradation can be estimated based on half-life and time of travel. The analysis of dilution is not critical when the variability associated with the toxicological information and the wastewater concentration estimate is considered. Using the mean annual river flow should be appropriate for most pollutants. If a pollutant is a fast-acting toxicant, the 30-day average low flow might be more appropriate.

If the available toxicological information is expressed in terms of an acceptable dose (milligrams per kilogram [mg/kg] body weight), the exposure must be calculated in terms of dose. The values normally used by EPA for estimation of waterborne exposure are the daily consumption of 2 liters of water and 6.5 g of fish, or other food derived from the affected area, by a 70 kg adult for the lifetime of the population. The value for food consumption can be adjusted for differences in diet if the pollutant is localized. A second scenario is used by the Office of Drinking Water in deriving Health Effects Advisories (formerly SNARLS). In order to protect the most sensitive group, exposure is determined using a 10 kg child drinking 1 liter of water per day.

Assessing the bioaccumulation route of exposure involves estimating fish body burden and human consumption. BCF can be calculated from the partition coefficient or may be available in the literature. As stated above, EPA typically uses a daily consumption of 6.5 g of fish by a 70 kg adult for 60 to 70 years. These values are used for national risk assessments and may be altered if local information is available. Impacts to wildlife should also be considered in evaluating the discharge of toxicants.

If the exposure exceeds levels that the toxicology information indicates as no effect levels, regulatory controls are needed. Discharge requirements are simply back-calculated from the exposure or dose level that is deemed acceptable using the same assumptions as were used in the assessment. How the acceptable exposure or dose was selected should be described in the permit fact sheet.

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5. *Exposure and Wasteload Allocation*

Introduction

At this point in the toxics control process, a water quality problem area has been identified. Screening analyses may have been done to assess the extent of toxicity, or the decision may have been made to proceed directly to a wasteload allocation (WLA). Criteria that will protect the receiving water have been determined from the following sources (see Section 2 and Appendix D):

- For specific toxicants, approved State standards (numerical or narrative) have been used. Site-specific criteria may have been derived with an EPA-approved methodology.
- For effluent toxicity, approved State standards have been used. Various species may have been tested for sensitivity to the toxicants, and criteria have been derived using an acute-chronic ratio (ACR) and species sensitivity factor as appropriate.
- For both specific toxicants and effluent toxicity, appropriate durations and frequencies for criteria compliance have been selected.

The next step is to assess the exposure of the resident aquatic community to toxic conditions. Exposure assessments involve predictions of how much of a waterbody is subject to concentrations exceeding water quality criteria, for how long, and how frequently. The spatial and temporal extent of aquatic life exposure to toxicants will vary depending on variations in the assimilative capacity of the receiving water and variations in effluent composition and quantity. WLA models are used to predict exposure and to calculate the effluent quality required to meet the criteria and protect the beneficial uses of the receiving water. This section describes the types of WLA modeling that can be performed. Data requirements for each of these modeling techniques are also described so that the definitive data generation procedures presented in Section 3 can be designed to support the specific type of WLA modeling a regulator chooses to perform.

Part 1 of this section discusses EPA's mixing zone policy and explains how this policy affects the allowable toxic load that can be discharged from a treatment plant. At present most WLA studies do not include mixing zone analyses since they concentrate on conventional pollutants that have their greatest impact downstream of the point of complete mixing. Mixing zone modeling has generally been performed only for discharges to large rivers, lakes, or estuaries where water quality problems are confined to the mixing area. Now that WLA studies will be conducted for toxicants, mixing analyses may be needed in more situations since many of these pollutants exert their maximum toxicity close to the point of discharge. Part 1 describes the tracer studies and mixing zone models that can be used to: 1) ensure that the discharge conforms to the State's allowable mixing zone dimensions; 2) prevent the mixing area from extending into critical resource areas; and 3) provide boundary conditions for the completely mixed WLA models discussed in Part 2 of this section.

Part 2 of this section describes the various types of models for completely mixed conditions that can predict the effect of wastewater discharges on the duration and frequency of toxicant concentrations in a receiving water. For steady state modeling of toxic discharges, Part 2 recommends the use of toxicologically based design flows for rivers and critical conditions for lakes and estuaries. As a replacement for steady state models, Part 2 also recommends the use of continuous simulation or probabilistic models to predict more accurately the duration and frequency with which criteria may be exceeded. These non-steady state models are appropriate for river, lake, and estuary analyses where adequate data are available.

Part 1 — The Mixing Zone and Toxic Wasteload Allocations

Overview

When wastewater is discharged into a waterbody, its transport may be divided into two stages with distinctive mixing characteristics. Mixing and dilution in the first stage is determined by the initial momentum and buoyancy of the discharge. The second stage covers a more extensive area in which the effect of initial momentum and buoyancy is overridden, and the waste is mixed primarily by ambient turbulence. In large rivers or estuaries, this second stage mixing area may extend for miles before uniformly mixed conditions are attained. In some instances, such as larger lakes or coastal bays, completely mixed conditions are never reached in the waterbody.

As shown in Table 5-1, most States specify spatial dimensions for regulatory mixing zones that do not

Table 5-1. State-by-State Mixing Zone Dimensions

State	Water Body	Dimensions
Alabama	0	0
Alaska	river, streams	≤ 1/3 CS
	lakes	≤ 10% SA
Arizona	streams	≤ 1/2 CS, < 500 meters length
	lakes	≤ 10% SA
Arkansas	large streams	≤ 1/4 CS
	small streams	≤ 2/3 CS
California	0	0
Colorado	0	0
Connecticut	streams	≤ 1/4 CS
Delaware	streams	≤ 1/3 CS
	lakes	≤ 10% SA
District of Columbia	estuary	≤ 10% CS
Florida	streams, rivers	≤ 800 meters ≤ 10% total length
	lakes, estuaries	≤ 125,600 m² (600′ radius) ≤ 10% SA
Georgia	0	0
Hawaii	streams	0
	estuaries	0
Idaho	0	0
Illinois	all	≤ 600 ft radius
	streams	≤ 1/4 CS
Indiana	streams	≤ 1/4 CS
Iowa	streams	≤ 1/4 CS
Kansas	streams	≤ 1/4 CS
Kentucky	streams	≤ 1/3 CS
Louisiana	streams	≤ 1/4 CS
Maine	streams	≤ 1/4 CS
Maryland	0	0
Massachusetts	0	0
Michigan	streams	≤ 1/4 CS
	Lake Michigan	≤ 1000 ft radius
Minnesota	streams	≤ 1/4 CS
Mississippi	0	0
Missouri	streams	≤ 1/4 CS
Montana	0	0
Nebraska	0	0
Nevada	streams	0
New Hampshire	streams	≤ 1/4 CS
New Jersey	streams	≤ 1/4 CS (thermal)
New Mexico	streams	≤ 1/4 CS
New York	streams	≤ 1/2 CS (thermal)
North Carolina	0	0
North Dakota	streams	≤ 1/4 CS
Ohio	receiving watercourse	≤ 1/3 CS
	mouth of receiving	≤ 1/5 CS
Oklahoma	streams	≤ 1/4 CS
Oregon	0	0
Pennsylvania	0	0
Rhode Island	streams	≤ 1/4 CS (thermal)
South Carolina	0	0
South Dakota	streams	≤ 3/4 CS or 100 yards of stream width
Tennessee	0	0
Texas	streams	≤ 1/4 CS
Utah	0	0
Vermont	streams	≤ 1/4 CS
Virginia	0	0
Washington	0	0
West Virginia	warm water fish streams	≤ 33% CS
	cold water fish streams	≤ 20% CS
	lakes	≤ 300′ any direction
Wisconsin	streams	≤ 1/4 CS
Wyoming	0	0
Guam	streams	≤ 1/4 CS, < 5 stream width length
Puerto Rico	streams	≤ 1/4 CS
	IMZ	≤ 400 ft
	FMZ	≤ 4000 ft
Virgin Islands	streams	≤ 1/4 CS

CS = cross-sectional area SA = surface area 0 = not listed

Source: Update of information in U.S. EPA Draft Technical Guidance Manual for the Regulations Promulgated Pursuant to Section 301(g) of Clean Water Act of 1977.

conform to the dimensions of the physically based mixing areas described above. State regulations dealing with streams and rivers generally limit mixing zone widths or cross-sectional areas and allow lengths to be determined on a case-by-case basis. In the case of lakes, estuaries, and coastal waters, dimensions are sometimes specified for the surface area that can be affected by the discharge. The surface area limitation usually includes the underlying water column and benthic area. State specification of mixing zone dimensions means that the WLA cannot be based on the assumption that the wastewater is completely mixed in the receiving water. State standards require that criteria be met at the edge of the [illegible]latory mixing zone 1) to provide a continuous zone of passage that meets water quality criteria for free-swimming and drifting organisms and 2) to prevent impairment of critical resource areas.

Even when State regulations include detailed geometric specifications for mixing zones, a smaller mixing zone or no zone at all can be required in site-specific cases. Bioaccumulative pollutants in particular may exert unexpected impacts if a mixing zone is allowed. A regulator should consider prohibiting a mixing zone in situations where an effluent is known to attract fish. In such cases, provision of a continuous zone of passage around the mixing area will not serve the purpose of protecting aquatic life. A review of the technical literature on avoidance/attraction behavior revealed that the majority of toxicants elicited an avoidance response or a neutral response [1]. The 53 toxicants in this review included a variety of heavy metals, organochlorine compounds, organophosphates, phenols, and resin acids. Six individual pollutants (copper, cadmium, mercury, total residual chlorine, chloroform, and NTA) were reported as eliciting attractive responses in one or more of the tested species. It should be noted that fish may not be attracted to effluent containing these attractive pollutants if other chemicals that cause avoidance behavior are present. The reverse is also true; an attractive pollutant or preferred temperature may overcome the avoidance reaction to specific chemicals in an effluent.

The purpose of this part of the Exposure Assessment Section is: 1) to describe EPA's mixing zone policy and 2) to summarize the monitoring and modeling techniques that are available for mixing analyses. In general, the modeling techniques are applicable to routine situations. Depending on the complexity of the hydrodynamics of the discharge and the receiving water, more detailed studies and/or the advice of experts may be required. The policy and techniques apply to both individual toxicants and effluent toxicity.

EPA Mixing Zone Policy

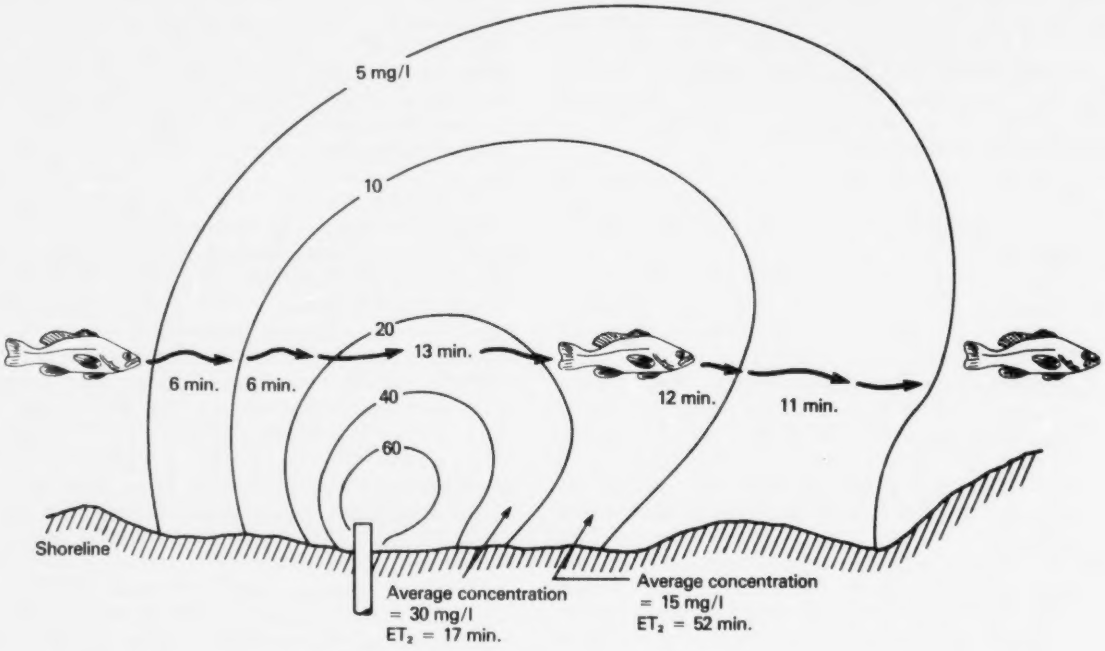
The EPA mixing zone policy is described in the 1983 Water Quality Standards Handbook [2]. The 1983 policy describes the mixing zone as an "allocated impact zone" where numeric water quality criteria can be exceeded as long as acutely toxic conditions are prevented. "Acutely toxic" conditions are defined as those conditions lethal to aquatic organisms passing through the mixing zone. Lethality is a function of the magnitude of pollutant concentrations and the duration of organism exposure to those concentrations.

EPA's water quality criteria provide guidance on the magnitude and duration of pollutant concentrations causing lethality. The criterion maximum concentration (CMC) for whole-effluent toxicity prevents lethality or acute effects in the tested species. The CMC for individual toxicants prevents acute effects in all but a small percentage of the tested species. Exposure to concentrations in excess of the CMC must be limited to minutes to prevent lethality or acute effects. Figure 5-1 illustrates how estimates of organism exposure time can be made by comparing swimming or drifting rates to the dimensions of concentration gradients within the regulatory mixing zone. Lethality can be prevented in the mixing zone in one of two ways. The first method is to prohibit lethal concentrations (values in excess of the CMC) in the pipe itself. The second approach is to use high rate diffusers and to ensure that the CMC is met within a short distance from the outfall. Details on how to use this second method are provided in the next subsection.

Proper design of a WLA study for a particular waterbody requires estimation of the distance from the outfall to the point where the effluent mixes completely with the receiving water. This determination can be based on the results of a tracer study or the simple desktop calculations described in the following subsections. If the mixing area is insignificant in size, as will occur with effluent-dominated streams, completely mixed conditions can be assumed, and the fate and transport models described in Part 2 can be used for the WLA.

At completely mixed conditions, two WLAs should be performed — one to meet the CMC and one to meet the criterion continuous concentration (CCC) at the required duration and frequency. Appendix D describes appropriate durations and frequencies for the CMC and CCC. As described in Part 2 of this section, both steady state and dynamic modeling can be performed to make these allocations. The more stringent of the two WLAs will determine the final permit limit.

If completely mixed conditions do not occur within a short distance of the outfall, the WLA study should rely on mixing zone monitoring and modeling. Just as in the case of completely mixed models, mixing zone analyses can be performed using both steady state and dynamic techniques. State requirements regarding the



Source: National Academy of Science. *Water Quality Criteria* 1972.

Figure 5-1. Hypothetical predicted concentrations of toxicant in an effluent plume and times of passage of migrating fish.

mixing zone will determine how water quality criteria are used in the WLA.

If a State allows a mixing zone, the WLA must ensure that, under chronic design conditions, concentrations at the mixing zone boundary meet the CCC. The CCC for whole-effluent toxicity is defined as a No Observed Effect Level (NOEL) that will prevent lethality and growth and reproduction impairment in the tested species. EPA's CCC for individual toxicants prevents unacceptable effects on the survival, growth, or reproduction of all but a small percentage of the tested species. If a State does not allow a mixing zone, concentrations in the pipe itself cannot exceed the CCC.

The following subsections describe how lethal conditions can be prevented in the mixing zone of each type of receiving water. General guidelines are provided on how to monitor mixing zone concentrations and how to design appropriate outlet structures. Some desktop and computer models for predicting mixing zone dimensions will also be summarized.

Design Criteria to Prevent Lethality in the Mixing Zone

In order to prevent lethal conditions in the regulatory mixing zone, the State can prohibit lethal concentrations in the pipe itself or require high rate diffusers and CMC compliance within a short distance of the outfall. If the high rate diffuser alternative is selected, the outfall discharge velocity must exceed three meters per second (ten feet per second). This will provide turbulent initial mixing and will minimize organism exposure time. In order to limit exposure time further, the outfall design must ensure that the most restrictive of the following conditions are met:

- The CMC must be met within 10% of the distance from the edge of the outfall structure to the edge of the regulatory mixing zone in any spatial direction.
- The CMC must be met within a distance of 50 times the discharge length scale in any spatial direction. The discharge length scale is defined as the square-root of the cross-sectional area of any discharge outlet. In the case of a multiport diffuser, this requirement must be met for each port using the appropriate discharge length scale of that port. This restriction will ensure a dilution factor of at least 10 within this distance under all possible circumstances, including situations of severe bottom interaction, surface interaction, or lateral merging.
- The CMC must be met within a distance of 5 times the local water depth in any horizontal direction from any discharge outlet. The local water depth is defined as the natural water depth (existing prior to the installation of the discharge outlet) prevailing under mixing zone design conditions (e.g., low flow for rivers). This restriction will prevent locating the discharge in very shallow environments or very close to shore, which would result in significant surface and bottom concentrations.

General Recommendations for Tracer Studies

Analysis of the mixing zone should be performed under critical conditions of minimal dilution. Some of those design conditions are summarized in the following subsections dealing with specific waterbodies. It should be noted, however, that occasionally certain off-design conditions may cause reduced mixing and water quality problems. Off-design conditions are described in the subsections below and should also be addressed in analyses.

Once the critical design condition has been selected for a waterbody, tracer studies or predictive models can be used to provide data on the dimensions and dilution of the wastewater plume during this critical period. The concentrations in the discharge jet or plume should be superimposed over a map of the various resource zones of the waterbody. The map will illustrate whether the State's required zone of passage is provided and whether the plume avoids critical resource areas. The WLA must be calculated to provide the appropriate zone of passage and to prevent detrimental impacts on spawning grounds, nurseries, water supply intakes, bathing areas, and other important resource areas.

A number of references that describe how to conduct tracer studies for mixing analyses are available [3-10]. For WLA studies in which a discharge is already in operation, tracer studies can be used to determine concentration isopleths in the mixing zone. This type of study inherently includes any discharge-induced (or first stage) mixing. However, if the outfall is not yet in operation, it is impossible to determine discharge-induced mixing by tracer studies. Tracer studies can be used in these situations to determine characteristics of the ambient (or second stage) mixing. For ambient mixing studies, the tracer release can be either instantaneous or continuous. Instantaneous releases are used frequently to measure longitudinal dispersion, but can also be used to determine lateral mixing in rivers [11] and lateral and vertical mixing in estuaries, bays, reservoirs, and lakes. For waterbodies with significant flow velocities, continuous releases of tracer are normally used to determine lateral and vertical mixing coefficients. Continuous releases can also be used to determine three-dimensional concentration isopleths for steady state conditions. The tracer study must be made at critical design conditions in order to use the results directly for WLAs. If a tracer study for ambient mixing is conducted at near-to-design conditions, the observed data can be used to determine dimensionless mixing coefficients. These coefficients can then be extrapolated to critical conditions using hydraulic parameters [11]. A tracer study at near-to-design conditions can also be used to determine the computer model required to predict critical condition mixing and to provide the coefficients needed for that WLA model.

General Recommendations for Outfall Design

Surface discharges at the shoreline of a waterbody are not recommended for toxic discharges. They usually have an impact along the shoreline when there is a significant crossflow, and they yield high surface concentrations.

Submerged discharges offer more flexibility in meeting the design goals for toxic discharges. They may be in the form of a single pipe outlet or of multiport diffusers, giving rise to one or several submerged discharge jets. Submerged multiport discharges do, however, have certain limitations in their use. They are not feasible in very shallow waterbodies or where periodic dredging or considerable scour and deposition occurs.

Specific design objectives for submerged discharges should be to avoid direct surface impingement and bottom attachment of the submerged jet or jets. Surface and bottom impacts should be evaluated at critical design conditions (low flow or high stratification) and at off-design conditions (higher flow or lower stratification) in order to ensure the best placement and design of the diffuser. Multiport diffusers provide more dilution than single outlets, but the alignment of the diffuser with the receiving water flow direction influences how much dilution will be provided. If the outlet structure is directed parallel to the direction of flow, dilution under high ambient velocities (off-design conditions) may be worse than under low velocities (critical design conditions). Some of the complexities of multiport diffusers have been summarized by Jirka [12], Roberts [13], and Holley and Jirka [11].

In rivers the preferred arrangement for a submerged discharge is to direct the outlet into the flow direction or vertically upward. In order to deal with the reversing currents of estuaries and coastal bays, the preferred arrangements for offshore discharges are parallel diffuser alignment (tee diffuser) or perpendicular diffuser alignment (staged diffuser) [12]. In lakes and reservoirs, the preferred arrangement for a negatively buoyant discharge is to direct the diffuser vertically upward. A positively buoyant, vertically directed jet could penetrate stratification, so the preference for this discharge is to orient the diffuser at a slight angle above the horizontal.

River and Run-of-river Reservoir Analyses

Rivers and run-of-river reservoirs are waterbodies that have a persistent through-flow in the downstream direction and do not exhibit significant natural density stratification. Interim recommendations on toxicologically based design flows for mixing zone and completely mixed steady state modeling of rivers are described in Appendix D of this document. Run-of-river reservoirs with residence times less than 20 days at critical conditions should also be analyzed using these design flows [14]. Regulated rivers may have a minimum flow in excess of these toxicological flows. In such cases, the minimum flow should be used in WLA modeling.

When a waste is discharged into a river, any discharge-induced mixing (i.e., mixing associated with the momentum and/or buoyancy of the effluent) occurs first. The second stage of mixing is controlled by ambient turbulence. The ambient mixing first accomplishes mixing over the depth, if this was not provided by the discharge-induced mixing, and then ambient mixing accomplishes mixing over the width. A final stage occurs after the waste is fully mixed throughout the cross-section. For nearly steady state conditions in rivers, the concentration distribution in this final stage is influenced primarily by the flow velocity and any biochemical and physico-chemical processes.

Models for Discharge-induced Mixing

The first stage of mixing is controlled by jet momentum and buoyancy. This stage generally encompasses most of the regulatory mixing zone. Especially in shallow environments, it is important to ascertain whether near field instabilities occur. These instabilities, associated with surface and bottom interaction and localized recirculation cells extending over the entire water depth, can cause build-up of effluent concentrations. Criteria for these instabilities and specialized predictive models have been developed [11,12].

In the absence of near field instabilities, horizontal or nearly horizontal discharges will create a clearly defined jet in the water column that will initially occupy only a small fraction of the available water depth. The following equations and models are designed for such stable near field conditions.

A minimum estimate of the dilution available in a mixing zone can be made using this equation:

$$S = 0.31 \frac{s}{D}$$

Where S = flux-averaged dilution
s = distance from outlet to edge of the mixing zone
D = diameter of outlet

The equation provides a *minimum* estimate of mixing zone dilution because it is based on the assumptions that ambient velocity is zero and that the discharge is neutrally buoyant. The equation also assumes that the jet is strong enough to reach the edge of the mixing zone. The same equation can be used to estimate the minimum dilution within the mixing zone before the CMC must be met to prevent lethal conditions.

Mixing graphs that include the effect of discharge buoyancy and ambient velocity can be found in Holley and Jirka [11], Fischer et al. [15], and Wright [16]. They are useful and accurate for design to meet criteria at the edge of the regulatory mixing zone.

More detailed design data for the mixing zone can be obtained from the use of computer models based on integral jet techniques. The models recommended here are available through the EPA. Other models can be used if well documented and verified with existing data. It is important to note that most models represent an idealization of actual field conditions and must be used with caution to ensure that the underlying model assumptions hold for the site specific situation being modeled. In general these buoyant jet models require the following input data: discharge depth, effluent flow rates, density of effluent, density gradients in receiving water, ambient current speed and direction, and outfall characteristics (port size, spacing, and orientation). Model output includes the dimensions of the plume at each integration step, time of travel to points along the plume centerline, and the average dilution at each point. The following models are available for use in riverine analyses.

MERGE is an ocean plume model that can be used for riverine analyses by setting ambient density gradients to zero. The model analyzes positively buoyant, coflowing multiple-port discharges [17]. Current speed is allowed to vary with depth. The multiple diffuser ports must be equally spaced and in line but may be oriented at any common elevation angle.

LINE is an ocean plume model that can be used for riverine analyses by setting ambient density gradients to zero. The model accounts for adjacent plume interference from a line source of positively buoyant, multiple-port discharges [18]. Unlike MERGE, diffuser alignment can be specified for LINE. Current speed is allowed to vary with depth.

OUTPLM is an ocean plume model that can be used for riverine analyses by setting ambient density gradients to zero. The model analyzes a positively buoyant, coflowing single-port discharge into a waterbody with a uniform ambient current [19, 20]. The model does not include the dynamics of surface impingement or the development of a bifurcated plume after impingement. A model that includes these features has been developed but is not available from EPA [21].

Mixing zone analyses for run-of-river reservoirs can use all the models listed under discharge-induced mixing for reservoirs, except for PLUME, which requires an ambient velocity of zero.

Models for Ambient Mixing

The models for discharge-induced mixing can be used to predict concentrations in the regulatory mixing zone when there is strong jet mixing. Beyond this point, the second stage of mixing in rivers is controlled by ambient turbulence. Pollutant concentrations at the end of this second zone must be predicted to provide boundary conditions for the completely mixed fate and transport models described in Part 2 of this section. Concentration distributions within the ambient mixing zone may also be needed to estimate concentrations at important resource areas or at downstream dischargers.

If there is no discharge-induced mixing associated with buoyant jet action of the discharge, then mixing over the depth of the river must be accomplished by ambient mixing. For a neutrally buoyant, soluble effluent discharged with low velocity at the surface or at the bed of a stream, the flow distance required to achieve complete vertical mixing is on the order of 50 to 100 times the depth of water in that portion of the channel where the effluent is discharged [22]. For a discharge that is either lighter (positively buoyant) or heavier (negatively buoyant) than the ambient water but still has no excess momentum, the flow distance for mixing over the depth will be greater. In the normal case with a high-velocity jet designed to prevent lethality in the mixing zone, mixing over the depth will be accomplished primarily by jet action, and the distance required for this vertical mixing will be much shorter.

In general, ambient mixing must also accomplish mixing over the width of a stream to bring the effluent to the completely mixed condition. For situations where the width of the zone that is mixed by the discharge-induced mixing is much smaller than the width of the river, the flow distance (X_m) required to achieve the completely mixed condition may be estimated from an equation of the form [15,11]:

$$X_m = \frac{mW^2u}{D_y}$$

where W = width of the river, u = flow velocity for the critical design flow, D_y = lateral dispersion coefficient as discussed in the next paragraph, and m = a parameter whose value depends on the degree of uniformity used to define "complete mixing" and on the transverse location of the outfall in the stream. If completely mixed conditions are defined as a 5% variation in concentration across the stream width, the value of m would be 0.3 to 0.4 for a discharge near the center of river flow (not the center of river width) and 0.4 to 0.5 for a discharge near the edge of the river. If, because of other uncertainties in the problem, a 25% variation across the width can be accepted as being completely mixed, then the corresponding ranges of values for m for the two locations would be 0.1 to 0.2 and 0.2 to 0.3. For a very small stream, X_m may be only a few hundred feet; for medium and large streams, X_m is normally several miles to several tens of miles. For diffusers that initially spread the discharge across a significant part of the river width or for cases where the discharge-induced mixing causes mixing across a significant part of the river width, the values of m and X_m can be smaller than the ones indicated here. For distances greater than X_m, the completely mixed models discussed in Part 2 of this section may be used. For shorter distances, the maximum concentrations in the river can be much greater than those predicted by completely mixed models and should be estimated using the following equations.

The lateral dispersion coefficient for most rivers can be calculated with the following equation [15, 11]:

$$D_y = 0.6\, du^* \pm 50\%$$

where d=water depth at critical design flow and u*=shear velocity. The coefficient (0.6) can vary between 0.3 and 1.0 depending on the type and degree of irregularity of the channel cross sections. Values approaching and exceeding 1.0 are normally associated with significant channel meandering [22]. The following equation for shear velocity should be used [15]:

$$u^* = (gdS)^{1/2}$$

where g=acceleration due to gravity, S=slope of the channel, and d=water depth.

Once an estimate of the distance to complete mixing is made with the above equations, the following equation can be used to predict concentrations upstream of this distance:

$$C_x = \frac{C_e\, Q_e\, W}{Q_s\, (\pi\, D_y\, X/u)^{1/2}}$$

where C_x=maximum pollutant concentration at distance x from the outlet, C_e=effluent concentration, Q_e=effluent flow, Q_s=critical design stream flow, X=distance from outlet, D_y=lateral dispersion coefficient, W=stream width, and u=flow velocity for the critical design flow.

It should be noted that this estimate of C_x is a worst case prediction since the equations assume no significant discharge-induced mixing and a neutrally buoyant effluent. A more accurate way to predict concentrations within this second stage of mixing is to use the methods of Yotsukura and Sayre [22]. In order to use this approach, however, the value of D_y and pollutant concentrations after discharge-induced mixing must be known from tracer studies and/or from the detailed modeling discussed in the previous subsection.

The PSY model can also be used to predict ambient mixing in shallow, freshwater streams where water depth is small in proportion to the width. PSY is a steady state two-dimensional plume model that predicts dilution of a surface discharge in a shallow receiving water where the plume attaches to both bottom and near shore [23]. Uniform vertical mixing is assumed to occur at the point of discharge.

Lake and Reservoir Analyses

This receiving water category encompasses lakes and reservoirs with residence times in excess of 20 days at critical conditions [14]. In the case of long and narrow reservoirs, areas above the plunge point can be analyzed as rivers and areas below as reservoirs [14]. Since effluent density relative to the ambient water can vary over seasons, no one season or stratification condition can be selected as the most critical dilution situation for all cases. In general, four seasons should be analyzed to determine the most critical period(s) for mixing zone analyses. All seasonal analyses should assume an ambient velocity of zero unless persistent currents have been documented. Special attention should be given to rising water level periods since pollutants can move back into coves and accumulate under these conditions. Location of discharges in coves and dead-end embayments should be prevented whenever possible.

Models for Discharge-induced Mixing

The same discharge-induced mixing equations and models available for riverine analyses are also appropriate to predict concentrations in the regulatory mixing zone of lakes and reservoirs. The only difference is that ambient velocity should always be set equal to zero unless persistent currents occur in a site-specific case. In addition, the following models are appropriate for lake and reservoir analyses:

PLUME analyzes a positively buoyant, single-port discharge into an arbitrarily stratified water body [24, 25]. No current is allowed in PLUME so stagnant conditions must prevail. Both surfacing and non-surfacing plumes can be traced with the model.

PDS is a steady state three-dimensional plume model that simulates the mixing of a surface discharge in a deep receiving water [26,27]. The plume is assumed to make no contact with the bottom or shoreline. Ambient current must be less than one-half the discharge velocity.

PDSM is a steady state three-dimensional plume model that predicts the dilution of a surface plume in a deep receiving water [27]. PDSM assumes that the plume is not attached to the bottom but that high ambient currents cause the plume to attach to the near shore.

DKHDEN is an ocean plume model that can be used for riverine analyses by setting ambient density gradients to zero. The model is designed for submerged, positively buoyant discharges at sufficient depth that the plume never surfaces [28]. The effects of a single or merging multiple-port diffuser can be simulated, but the outlets must be directed parallel to the flow direction (co-flowing diffuser design). Current speed is allowed to vary with depth.

Models for Ambient Mixing

Discharge-induced mixing is especially crucial for lakes and reservoirs as ambient mixing processes play a lesser role. For WLA modeling purposes, the discharge-induced mixing concentrations can be used as boundary conditions for two-dimensional models that are appropriately configured for this input. These models are referenced in Part 2 of this section.

Estuary and Coastal Bay Analyses

This receiving water category encompasses estuaries, which are defined as having a main channel reversing flow, and coastal bays, which are defined as having significant two-dimensional flow in the horizontal directions. For both waterbodies, the critical design conditions recommended here are based on astronomical, not meteorological, tides.

In estuaries without stratification, the critical dilution condition includes a combination of low water slack at spring tide for the estuary and design low-flow for riverine inflow. In estuaries with stratification, a site-specific analysis of a period of minimum stratification and a period of maximum stratification, both at low-water slack, must be made to evaluate which one results in the lowest dilution. In general, minimum stratification is associated with low river inflows and large tidal ranges (spring tide), whereas maximum stratification is associated with high river inflows and low tidal ranges (neap tide).

After evaluating either stratified or unstratified estuaries at critical design conditions, an off-design condition should be checked. The off-design condition recommended for both cases is the period of maximum velocity during a tidal cycle. This off-design condition results in greater dilution than the design condition, but it causes the longest extension of the plume. Extension of the plume into critical resource areas may cause more water quality problems than the high concentration, low dilution situation.

Recommendations of a critical design for coastal bays are the same as for stratified estuaries. The period of maximum stratification must be compared with the period of minimum stratification in order to select the worst case. The off-design condition of maximum tidal velocity should also be evaluated to predict the extent of the plume.

Models for Discharge-induced Mixing

Most of the discharge-induced mixing equations and models recommended for riverine and reservoir analyses are also appropriate to predict concentrations in the regulatory mixing zone of estuaries and coastal bays.

A minimum estimate of the dilution available in a regulatory mixing zone can be made with the same equation used previously:

$$S = 0.31 \frac{s}{D}$$

Mixing graphs that include the effect of discharge buoyancy, ambient velocity, *and* stratification can be found in Fischer et al. [15] and Wright [16].

Models for Ambient Mixing

For estuaries that are completely mixed with regard to salinity, the equations presented for rivers can be used to estimate concentrations between the outlet and the point of complete mixing. These equations will be applicable to only a very few unstratified estuaries, however, since the time required to mix across the estuary must be significantly less than: 1) the time required for the effluent to pass out of the unstratified part of the estuary, 2) the time required for the effluent to pass into a segment of greatly changed cross-section, or 3) the time required for the substance to decay.

$$X_m = \frac{mW^2u}{D_y}$$

$$Dy = 0.6\ du^* \pm 50\%$$

$$u^* = 0.10u_t$$

$$C_x = \frac{C_e\ Q_e\ W}{Q_s\ (\pi\ D_y\ X/u)^{1/2}}$$

where all the variables are defined as for rivers except that u in the equation for X_m and C_x should be the velocity associated with the freshwater flow into the estuary, whereas u_t in the equation for u^* should be based on an average total velocity (including the tidal component).

For WLA modeling purposes, two-dimensional models can be configured to use discharge-induced mixing concentrations as input. These models are referenced in Part 2 of this section.

Ocean Analyses

Critical design conditions and modeling and monitoring techniques for oceans are described in the 301(h) Technical Support Document and the 301(h) publication entitled Initial Mixing Characteristics of Municipal Ocean Discharges [29,17]. This information will not be repeated here.

Part 2 — Wasteload Allocation Modeling

Overview

At the present time, most States and EPA Regions use steady state models that assume the wastewater is completely mixed with the receiving water in order to calculate WLAs for pollutants. As explained in Part 1 of this section, the assumption of completely mixed conditions may be adequate in most situations involving conventional pollutants since they have the greatest impact downstream of the discharger. Now that WLAs will be conducted for toxicants, mixing analyses may be needed in more situations because many of these pollutants exert maximum toxicity close to the point of discharge. The completely mixed models described here in Part 2 are appropriate for any effluent-dominated receiving waters or situations in which mixing is relatively complete within a short distance. In multiple-discharge situations, these models will also be needed to link the concentrations predicted at the edge of allowable mixing zones to downstream impacts.

The purpose of this part of the Toxics Control Support Document is to improve existing toxicant modeling procedures by: 1) recommending for steady state modeling of rivers the use of critical flows that are based on toxicological concerns and 2) recommending dynamic modeling techniques for rivers, lakes, and estuaries that are better suited to toxic exposure assessments than the steady state approach. Section 6 will describe how to improve the procedure for determining permit limits from the WLA results.

Steady State Modeling for Rivers, Lakes, and Estuaries

A steady state model requires single, constant inputs for effluent flow, effluent concentration, background receiving water concentration, receiving water flow, and meteorologic conditions such as temperature. The frequency and duration of ambient concentrations predicted with a steady state model must be assumed to equal the frequency and duration of the critical receiving water conditions used in the model. The variability in effluent flows and concentrations also affects receiving water concentrations, but these effects cannot be predicted with constant input. Steady state models can be improved for toxic WLAs by: 1) using design flows that will ensure criteria compliance at the appropriate duration and frequency and 2) calculating both acute and chronic WLAs.

Appendix D of this document describes the toxicological basis for selecting receiving water flows for steady state modeling. The duration of the design flow is based on the maximum exposure time that will prevent acute or chronic effects. The duration of flow is assumed to apply to the duration of the allowable effluent concentration or load. For example, if the flow used is a seven-day average value, the allowable load is considered to be a seven-day average. The return period or frequency of the flow is based on the number of years required for biological population recovery after criteria have been exceeded.

As an interim measure, Appendix D recommends specific flows for steady state modeling of rivers. These flows are suggested for use only until a detailed guidance manual is developed on how to select toxicological flows using site-specific data. At the present time, there are no recommended toxicological flows for steady state modeling of lakes, reservoirs, or estuaries. The critical conditions recommended for these waterbodies in Part 1 are worst case situations that are not derived from toxicological duration and frequency data. These worst case conditions should be used until further guidance is provided.

Another improvement in steady state toxics modeling can be realized by performing two WLAs — one for maximum criteria and one for continuous criteria. EPA is encouraging the States to adopt two-number water quality criteria that specify a CMC and a CCC. The CMC and CCC have recommended durations and frequencies associated with them (See Appendix D). Steady state WLA models should be used to calculate the allowable effluent load that will meet the maximum criterion at the acute design flow and the allowable load that will meet the continuous criterion at the chronic design flow. Calculation of these values will ensure that the treatment system is designed to comply with both criteria. Calculation of two WLAs will also provide needed data on effluent variability. This information will aid in the statistical derivation of permit limits that is discussed in Section 6.

Dynamic Modeling Techniques for Rivers, Lakes, and Estuaries

As explained in Appendix D, the extent of biological impairment from toxic discharges depends on the duration of criteria violations as well as the number of times these violations occur. The duration and frequency of violations depend, in turn, on the daily variation in receiving water and effluent flow combined with daily variation in effluent toxicity. Although the steady state modeling approach can only consider the impact of receiving water variability on criteria violations, dynamic modeling techniques can predict the effects of both receiving water and effluent variability. These more thorough methods calculate an entire probability distribution for receiving water concentrations rather than a single worst case based on critical conditions. The prediction of complete probability distributions allows the risks inherent in alternative treatment strategies to be quantified. The use of probability distributions in place of worst case conditions has been accepted practice for years in water resource engineering where it was found to produce more cost-effective design of bridge openings, channel capacities, flood plain zoning, and water supply systems. The same cost effectiveness can be realized for pollution controls if probability analyses are used. For example, comparison of receiving water concentration probability distributions can be used to determine which treatment system causes criteria to be exceeded less frequently — a less expensive system producing a higher mean toxicity concentration but more stable effluent versus a more expensive facility producing a lower average concentration but higher-variability effluent.

The dynamic modeling techniques have an additional advantage over steady state modeling in that they determine the entire effluent concentration distribution required to produce the desired frequency of criteria compliance. Maximum and monthly average permit limits can then be obtained directly from this distribution. Existing practice has been to use a steady state model to calculate only a chronic WLA. This generates a single allowable effluent value and no information about effluent variability. If the steady state model is used to calculate both acute and chronic wasteloads, limited information on variability will be provided, but the entire effluent distribution will not be predicted. As described in Section 6, steady state WLA values are more difficult to use in permits and enforcement.

The remainder of this subsection describes the dynamic methods, which include continuous simulation models, the lognormal probabilistic dilution model, and Monte Carlo simulation using steady state or continuous simulation models. Recommendations on when to apply these methods are also presented.

Continuous Simulation Models

As shown in Figure 5-2, a continuous simulation model uses each day's effluent flow (Q_e) and concentration data (C_e) with each day's receiving water flow (Q_s) and background concentration (C_s) to calculate downstream receiving water concentrations [30]. The model predicts these concentrations in chronological order with the same time sequence as the input variables (C_b vs. time). The daily receiving water concentrations can then be ranked from the lowest to the highest without regard to time sequence. A probability plot can be constructed from these ranked values, and the occurrence frequency of any one-day concentration of interest can be obtained (C_b vs. frequency). Running average concentrations for four days, or for any other averaging period, can also be computed from the daily concentrations, ranked in order of magnitude, and presented as a probability plot (see Figure 5-3).

The probability plot generated by the continuous simulation model using existing effluent data will indicate whether criteria are predicted to be exceeded more frequently than desired. Appendix D discusses how to select the appropriate occurrence frequency (or return period) based on the biological recovery period required for a specific waterbody. If recurrence intervals of 10 or 20 years are desired, at least 30 years of flow data should be available in order to provide a long enough record to estimate the probability of such rare events. The same data requirements are also true for the lognormal probabilistic and Monte Carlo methods.

If the existing effluent is predicted to cause too frequent instream violations of water quality criteria, additional treatment is necessary to protect the receiving water. Two possible approaches can be taken to determine the allowable effluent distribution. One approach is based on the simplifying assumption that treatment will only change the magnitude of effluent concentrations. No changes are assumed to occur in effluent flows or in the relative variability of effluent concentrations. With these assumptions, no additional model runs are needed to determine the allowable distribution for effluent concentrations. The new required effluent concentration distribution is assumed to be the same as the existing distribution except that it is reduced in magnitude by whichever is greater — the percentage necessary for the one-day average concentrations to meet the CMC or the four-day average concentrations

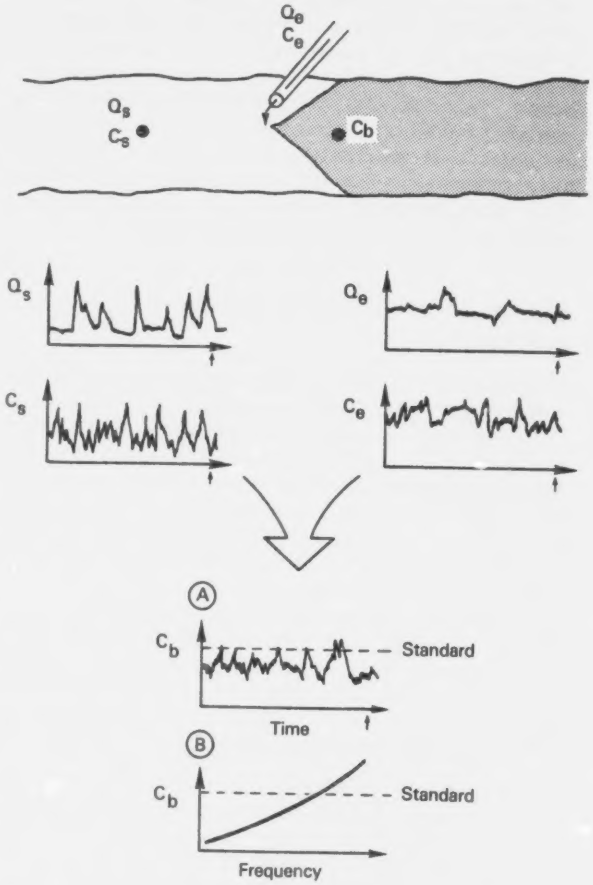


Figure 5-2. Continuous simulation modeling schematic.

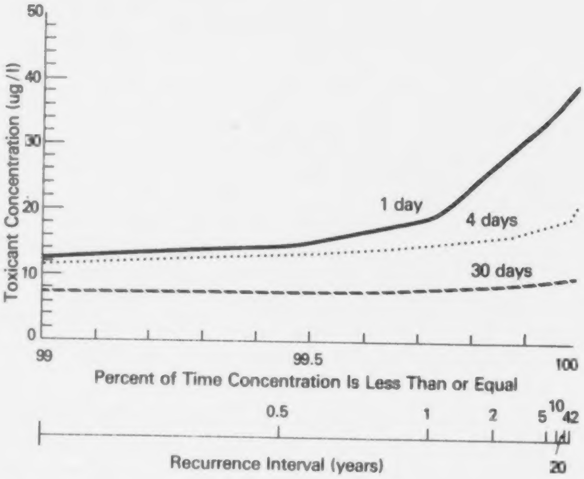


Figure 5-3. Concentration-frequency curves.

to meet the CCC at the desired recurrence interval. Section 6 includes the details on how permit limits are derived from the mean and coefficient of variation of effluent concentrations determined from this analysis. Information concerning effluent concentration means and variability can be obtained from databases on existing treatment plants and from Effluent Guideline documents.

It may not be valid to assume that effluent concentrations after treatment will have the same coefficient of variation as concentrations before treatment. (The coefficient of variation is the standard deviation of a parameter divided by its mean value.) Studies have documented that advanced secondary treatment increases the coefficient of variation of BOD and TSS concentrations compared to secondary treatment. Investigations must be conducted to determine how treatment processes for heavy metals, organic chemicals, and effluent toxicity will change the variability of these constituents. To account for a change in variability, an alternative approach must be used to determine the allowable effluent distribution. Iterative model runs will have to be performed using different concentration means with the effluent "future treatment" variance until a mean is found that meets the criteria at the desired recurrence intervals. These iterative model runs will require stochastic generation of effluent input data since daily effluent concentrations will not be available for the hypothetical treatment schemes. The required "future treatment" mean and coefficient of variation of effluent concentration can then be used as described in Section 6 to set permit limits.

EPA's Monitoring and Data Support Division (MDSD) has developed an interactive program that will automatically create daily effluent flow and concentration data for input to continuous simulation models. The program is included in the DYNTOX portion of the ANNIE program available from the EPA Modeling Center at the Athens, Georgia, Environmental Research Laboratory (ERL) [31]. If the observed data base is fairly complete but missing a few points, a linear interpolation scheme is used to fill in the gaps. If data are scarce, a lag-one Markov method is used to generate daily data stochastically. The lag-one Markov method uses the mean, standard deviation, and daily correlation coefficient of the observed data to create random sequences of data all having the same statistical properties. The interactive program is written in FORTRAN and is available for use on mainframe or IBM PC-compatible computers.

Continuous simulation models have the following advantages:

- They can predict the frequency and duration of toxicant concentrations in a receiving water.
- They include fate processes for specific toxicants.
- They include transport processes for rivers, lakes, and estuaries.
- They are capable of analyzing single or multiple pollutant sources.
- They incorporate the cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters.
- They incorporate the effect that the serial correlation of daily flows and other parameters has on the persistence of criteria violations.
- They are capable of synthesizing long term streamflow records for ungauged rivers using precipitation and evapotranspiration data.
- Their long simulation times prevent the initial conditions used in the model from affecting the calibration of fate and transport processes.

Continuous simulation models have the following disadvantages:

- They require more input data and calibration/verification data than a steady state model.
- They require timeseries input data.

Lognormal Probabilistic Model

Without resorting to the continuous simulation method of computing time variable receiving water concentrations, this probabilistic method uses the lognormal probability distributions of input variables to calculate probability distributions of output variables [32]. As a result, the method requires only the relevant statistical parameters of the input variables (medians and coefficients of variation [CVs]) rather than the actual timeseries data needed for continuous simulation. If one-day average receiving water concentrations must be predicted for comparison with the CMC, lognormal probability distributions of daily input data are needed. If four-day average concentrations must be predicted, the lognormal probability distributions of four-day average input data are required. Because this probabilistic model as yet cannot incorporate fate and transport processes, it can only be used to predict the concentration of a substance after complete mixing in a river and before decay or transformation significantly alters the concentration.

Two methods are available to compute the probability that downstream toxicants (or effluent toxicity) will exceed criteria — an approximate method of moments and numerical integration. The computations for both procedures can be programmed for desktop personal computers. EPA's MDSD has developed an interactive program that will perform the numerical integration calculations for the lognormal probabilistic dilution model. The program is included in the DYNTOX portion of the ANNIE program available from the EPA Modeling Center at the Athens, Georgia, ERL [31]. The interactive program is written in FORTRAN and is available for use on mainframe or IBM PC-compatible computers.

The method of moments will be discussed here. It is based on the following equation:

$$C_T = C_e \varnothing + C_s (1 - \varnothing)$$

where C_T = downstream concentration
C_e = effluent concentration
C_s = upstream concentration
$\varnothing$ = effluent flow fraction $(Q_e/(Q_e+Q_s))$

Estimates of the mean and variance of effluent concentration, effluent flow, and upstream concentration can be made by regressing the natural log of each of these variables against Z, the quantiles of a standard normal random variable. A three-parameter lognormal method can be used for streamflows since the two-parameter distribution may produce a poor fit for the low-flow tail. This problem is due to the fact that some streams have a base flow that does not fluctuate appreciably.

Once the logarithmic means and standard deviations of effluent flow and upstream flow are computed from the regression least squares fit, the distribution of the dilution factor ($\varnothing$) can be computed. Assuming that $\varnothing$, $1-\varnothing$, C_e, and C_s are lognormally distributed, the products $C_e\varnothing$ and $C_s(1-\varnothing)$ will also be lognormal. Even though it is not true that the sum of two lognormal variables is lognormal, this assumption must be made to proceed with the method. The method of moments also requires that effluent and upstream concentrations be uncorrelated with flow since no convenient moment approximations are available to account for cross correlation. The numerical integration method does include a correlation coefficient and this is the method used in the DYNTOX interactive program. The arithmetic moments of C_e, C_s, and $\varnothing$ can be back calculated from the logarithmic values and used to compute the arithmetic moments of the downstream concentration (C_T) as follows:

$$u(C_T) = u(C_e)\,u(\varnothing) + u(C_s)\,[1-u(\varnothing)]$$

$$\begin{aligned}\sigma^2(C_T) = {} & \sigma^2(\varnothing)\,[u(C_e)] - u(C_s)]^2 \\ & + \sigma^2(C_e)\,[\sigma^2(\varnothing) + u\,\sigma^2(\varnothing)] \\ & + \sigma^2(C_s)\,[\sigma^2(\varnothing) + (1-u(\varnothing))^2]\end{aligned}$$

The log mean and variance of the downstream concentration can then be calculated and a lognormal probability distribution created. The concentration at any recurrence interval of interest can be obtained from this distribution and compared to acute and chronic criteria.

The approaches for calculating the allowable effluent distribution using the lognormal probabilistic model are the same as those described for the continuous simulation model. The assumption can be made that treatment will only change the magnitude of concentrations, not their CV. The new required effluent concentration distribution is then assumed to be the same as the existing distribution except that it is reduced in magnitude by whichever is greater — the percentage necessary for the one-day average concentration to meet the CMC or for the four-day average concentrations to meet the CCC at the desired recurrence intervals. If it is known that the CV of effluent concentrations will change after additional treatment, iterative model runs will have to be performed using different concentration means with the appropriate "future treatment" variance until a mean is found that meets the CMC and CCC at the desired recurrence intervals. Whichever approach is used in the WLA, the procedures described in Section 6 can be used to set permit limits. Information concerning effluent concentration means and variability can be obtained from databases on existing treatment plants and from Effluent Guidelines documents.

The lognormal probabilistic dilution model has the following advantages:

- It can predict the frequency and duration of toxicant concentrations in a river.
- It does not require timeseries data.
- It can incorporate the cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters if the analysis is developed separately for each season and the results combined.

The lognormal probabilistic dilution model has the following disadvantages:

- It requires more input data than a steady state model.
- It does not include instream fate processes.
- It applies to rivers only; no lake or estuarine analyses are possible.
- It analyzes multiple pollutant sources inaccurately.
- It requires model input data to be lognormally distributed.

Monte Carlo Simulation

Monte Carlo simulation combines probabilistic and deterministic analyses since it uses a fate and transport mathematical model with statistically described inputs. Monte Carlo simulations have been the most frequently used approach in stochastic water quality studies [33-37]. The probability distributions of effluent flow, effluent concentration, and other model input must be defined using the appropriate duration for comparison to the CMC and CCC. If one-day average receiving water concentrations must be predicted for CMC comparisons, probability distributions of daily model input data are needed for Monte Carlo simulation. If four-day average concentrations must be predicted for CCC comparisons, the probability distributions of four-day average input data are required. The computer selects input values from these distributions using a random generating function. The fate and transport model is repetitively run for a large number of randomly selected input datasets. The result is a simulated sequence of receiving water concentrations. These concentrations do not follow the temporal sequence that is calculated with the continuous simulation model, but they can be ranked in order of magnitude and used to form a frequency distribution. Monte Carlo analyses can be used with steady state or continuous simulation models [38].

The approaches for calculating the allowable effluent concentration distribution using Monte Carlo simulation are the same as those described for the continuous simulation model. The assumption can be made that treatment will only change the magnitude of concentrations, not their CV. The new required effluent concentration distribution is assumed to be the same as the existing distribution except that it is reduced in magnitude by whichever is greater — the percentage necessary for the one-day average concentrations to meet the CMC or for the four-day average concentrations to meet the CCC at the desired recurrence intervals. If it is known that the coefficient of variation of effluent concentrations will change after additional treatment, iterative model runs will have to be performed using different concentration means with the appropriate "future treatment" variance until a mean is found that meets the acute and chronic criteria at the desired recurrence intervals. Whichever approach is used in the WLA, the procedures described in Section 6 can be used to set permit limits. Information concerning effluent concentration means and variability can be obtained from databases on existing treatment plants and from Effluent Guidelines documents.

EPA's MDSD has developed an interactive program that will randomly select the sets of input data required for Monte Carlo simulations. The program is included in the DYNTOX portion of the ANNIE program available from the EPA Modeling Center at the Athens, Georgia, ERL [31]. The interactive program is written in FORTRAN and is available for use on mainframe or IBM PC-compatible computers.

Monte Carlo simulation has the following advantages:

- It can predict the frequency and duration of toxicant concentrations in a receiving water.
- It can be used with steady state or continuous simulation models that include fate processes for specific toxicants.
- It can be used with steady state or continuous simulation models that include transport processes for river, lake, and estuary predictions.
- It can be used with steady state or continuous simulation models that are designed for single and multiple pollutant source analyses.
- It can estimate the uncertainty in model predictions.
- It does not require timeseries data.
- It does not require model input data to follow a specific statistical function.
- It can incorporate the cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters if the analysis is developed separately for each season and the results combined.

Monte Carlo simulation has the following disadvantage: it requires more input data and calibration/verification data than a steady state model.

Calculating the Return Period

Two common methods exist to calculate the return period for a given concentration from probabilistic modeling. Herein, for convenience, they are termed: 1) the percentile method and 2) the extrema method. The percentile method uses a listing of all individual daily concentrations and ranks them. The return period for a concentration is then calculated based on the percentile occurrence. In the extrema method, only annual extreme values are used in the ranking. The return periods calculated from these two methods are equally valid statistical representations. When using the percentile method, results express an average return period and include multiple occurrences within any year. On the other hand, the extrema method describes the return period for an annual extreme and includes only the extreme of multiple occurrences within any year. The DYNTOX program uses the percentile method to calculate return periods.

General Recommendations for Model Selection

The reliability of the predictions from any of the modeling techniques depends on the adequacy of the data used in the analyses. The minimum data required for model input includes receiving water flow/volume, effluent flow, effluent concentrations, and background concentrations. In many locations, streamflow data should be sufficient for both steady state and dynamic river models. At least 30 years of flow data are required if violations of the CMC and CCC must be evaluated at rare recurrence intervals of once in 10 or 20 years. Measurements of effluent toxicity or individual toxicants can be much more limited. Table 5-2 indicates the effect of sample size on the confidence with which a modeler can assume that the largest and smallest measurements bracket 75%, 90%, 95%, or 99% of the toxic concentrations discharged from a pollutant source. According to Table 5-2, which is based on order statistics, 30 effluent toxicity measurements are needed to provide 100% confidence that the range of the sampling data covers 75% of the actual toxicity. Twelve samples result in 84% confidence that 75% of the actual toxicity is represented by the data. This table can be used as a guideline to help determine how many measurements should be made for a reasonable estimate of effluent variability.

If only a few toxicant or effluent toxicity measurements are available, Level 1 or 2 steady state assessments should be used (See Table 5-3). Modeling should also be limited to steady state procedures if a daily receiving water flow record is not available. If State regulations do not require a specific design flow for river WLAs, toxicologically based design flows can be used to calculate the allowable effluent load. Appendix D describes how to select these design flows. Fate and transport models or dilution calculations can be used for individual toxicants. At the present time, only dilu-

Table 5-2. Confidence Associated With a Tolerance Limit Statement

n	P = .75	P = .90	P = .95	P = .99
3	.16	.03	.01	.00
4	.26	.05	.01	.00
5	.37	.08	.02	.00
6	.47	.11	.03	.00
7	.56	.15	.04	.00
8	.63	.19	.06	.00
9	.70	.23	.07	.00
10	.76	.26	.09	.00
11	.80	.30	.10	.01
12	.84	.34	.12	.01
13	.87	.38	.14	.01
14	.90	.42	.15	.01
15	.92	.45	.17	.01
16	.94	.49	.19	.01
17	.95	.52	.21	.01
18	.96	.55	.23	.01
19	.97	.58	.25	.02
20	.98	.61	.26	.02
25	.99	.73	.36	.03
30	1.00	.82	.45	.04
40	—	.92	.60	.06
50	—	.97	.72	.09
60	—	.99	.81	.12
70	—	.99	.87	.16
80	—	1.00-	.91	.19
90	—	—	.94	.23
100	—	—	.96	.26

Confidence γ with which we may assert that 100 P percent of the population lies between the largest and smallest of a random sample of n from that population (continuous distribution assumed).

Adapted with permission from *Annals of Mathematical Statistics,* Vol. 29, No. 2., June 1958, pp. 599-601, from article entitled "Tables for Obtaining Non-Parametric Tolerance Limits" by Paul N. Somerville.

Source: Natrella, M.G. *Experimental Statistics.* National Bureau of Standards Handbook 91, U.S. Department of Commerce, August 1963.

Table 5-3. Exposure Assessments

Option	Type of Model	Comments
Level 1	Steady state	7Q10 or other State required low flow
Level 2	Steady state	Toxicologically based design flow
Level 3	Continuous simulation	Batch process or seasonal effluent time series data
	Monte Carlo or Lognormal probabilistic	No batch process or seasonal trends in effluent data Effluent data only defined by statistical parameters

tion calculations or first order decay equations are recommended for effluent toxicity analyses. Section 1 discusses the conservative/additive assumption for toxicity.

If adequate receiving water flow and effluent concentration data are available to estimate their frequency distributions, one of the non-steady state modeling procedures should be used to develop more cost effective treatment requirements. If the effluent data exhibit significant seasonal differences or batch process trends, the Level 3 continuous simulation approach may be the easiest dynamic modeling method to use. The best results will, of course, be obtained if daily effluent flows and concentrations are available for model input for an entire year. The lag-one Markov technique can be used to generate daily effluent data for the entire simulation period as long as adequate measurements for the site specific plant (or a similar one) are available to estimate a day-to-day correlation coefficient and to determine when seasonal or batch process changes in effluent quality occur. The DYNTOX program described in the continuous simulation subsection can be used for this Markov generation of data.

If adequate receiving water flow and effluent concentration data are available and if effluent data exhibit no seasonal or batch process trends, the Level 3 lognormal and Monte Carlo methods may be easier and require less computer time than the continuous simulation approach. The DYNTOX program described in the sections on these modeling techniques can be used to perform the analyses.

Effluent Toxicity Modeling

In order to apply the steady state, continuous simulation, or probabilistic methods to effluent toxicity modeling, the percent effluent measurements should be converted to toxic units (TUs). A TU is the reciprocal of the dilution that produces the test endpoint. An acute toxic unit (TU_a) is the reciprocal of the effluent dilution that causes fifty percent of the organisms to die by the end of the acute exposure period. A chronic toxic unit (TU_c) is the reciprocal of the dilution that causes no unacceptable effect on the organisms by the end of the chronic exposure period. The WLA must ensure that the CMC and the CCC are met in the receiving water at the desired duration and frequency. The CMC is recommended as one-third the LC50 value. Thus the CMC for toxicity measured with acute tests would be equal to the following:

$$CMC = 0.3\ TU_a$$

The CCC for toxicity measured with chronic tests is recommended as the following:

$$CCC = 1.0\ TU_c$$

The first step in the WLA process is to calculate the allowable acute effluent toxicity that meets the CMC in the receiving water at the duration and frequency discussed in Appendix D. If effluent toxicity measurements for model input are based on acute toxicity tests using fewer than three species, it is recommended the CMC be divided by a factor of 10 to account for the uncertainty in sensitivity (CMC = 0.03 TU_a).

The next step in the WLA process is to calculate the allowable chronic effluent toxicity that meets the CCC in the receiving water at the duration and frequency discussed in Appendix D. It is recommended that the CCC be divided by the species sensitivity factor of 10 if measurements are based on tests of fewer than three species (CCC = 0.1 TU_c). In order to compare the allowable acute toxicity value to the allowable chronic toxicity value, the numbers must be converted to the same unit using the following relationship: 1 TU_a = (ACR)(TU_c), where the ACR is determined from tests on the effluent. It is important that the ACR used for WLA purposes be based on actual data and not assumed to be 10 or 20 as in the screening procedure (Section 3). The value of this ratio will influence whether the acute or chronic WLA is more stringent and is used to calculate the permit limit using Section 6 methods.

The following subsections recommend models for toxicity and individual toxicants for each type of receiving water — rivers, lakes, and estuaries. Detailed guidelines on the use of fate and transport models for individual toxicants are included in the WLA guidance available from EPA's MDSD [39-41].

Specific Model Recommendations — Rivers and Run-of-river Reservoirs

EPA's WLA guidance for modeling individual toxicants in streams and rivers recommends the following steady state models [39]: Simplified Lake/Stream Analysis (SLSA), Michigan River Model (MICHRIV), Chemical Transport and Analyses Program (CTAP), Exposure Analysis Modeling System (EXAMS), and Metals Exposure Analysis Modeling System (MEXAMS). All these models except MEXAMS can simulate both organic chemical and heavy metal fate and transport in rivers. MEXAMS is designed to predict the ionic speciation of metals only. All of these models are suitable for analysis of multiple reach, multiple pollutant source discharges, except for SLSA, which simulates a single reach and a single pollutant source. The DYNTOX program discussed in the dynamic modeling subsections can be used to develop input data to run these steady state fate and transport models with Monte Carlo techniques to predict instream concentration-frequency curves.

EPA's WLA guidance recommends the following continuous simulation models for individual toxicants in streams and rivers: Estuary and Stream Quality Model (WASTOX), Chemical Transport and Fate Model (TOXIWASP), Channel Transport Model (CHNTRN), Finite Element Transport Model (FETRA), Sediment Contaminant Transport Model (SERATRA), Transient One-dimensional Degradation and Migration Model (TODAM), and Hydrologic Simulation Program—Fortran (HSPF). All of these models are designed for multiple reach, multiple pollutant source analyses of both organic chemicals and heavy metals. The DYNTOX program discussed in the dynamic modeling subsections can be used to develop the input data for these continuous simulation fate and transport models to predict instream concentration-frequency curves. In addition the lognormal probabilistic dilution model, included in DYNTOX can be used for single discharge dilution analyses to predict concentration-frequency curves.

Several references that list reaction rates for the various fate processes affecting organic chemical and heavy metals are available [42-44]. At the present time, the fate of effluent toxicity in a receiving water is not understood. Even if a decay rate for toxicity can be measured on a given day in a site-specific situation, there is no way as yet to know how this rate is affected by temperature, pH, or other environmental conditions. There is also no way to know how this rate may change when new treatment is installed. Instream measurements of toxicity should be made at least once per season in order to identify any time-varying trends in site-specific fate processes. These monitored decay rates can then be used in steady state or continuous simulation fate and transport models to predict receiving water toxicity, assuming that the rates will not change with future treatment.

If instream toxicity measurements are not available, effluent toxicity cannot really be modeled. EPA ERL–Duluth has begun a three-year study of the persistence of toxicity that should eventually improve the modeling of toxicity. As an interim measure, it is recommended that effluent toxicity modeling be limited to dilution calculations and assumptions that toxicity is additive and conservative. Toxicity is expected to be additive even when the toxicity of one effluent affects invertebrates and the toxicity of a downstream discharge affects fish or algae. For rivers and run-of-river reservoirs with a detention time less than 20 days, the following dilution equation should be used assuming completely mixed conditions:

$$C = \frac{C_sQ_s + C_eQ_e}{Q_e + Q_s}$$

where C = downstream concentration (TU_c or TU_a)
C_s = upstream concentration (TU_c or TU_a)
Q_s = upstream flow (cfs)
C_e = effluent concentration (TU_c or TU_a)
Q_e = effluent flow (cfs)

For multiple dischargers, this equation must be applied sequentially to find the concentration as a function of distance downstream. The equation can be used for a steady state analysis if Q_s is set equal to the design flow, Q_e is set equal to the design plant flow, and C_e is calculated to meet the CMC and CCC. This equation can also be used with the continuous simulation, lognormal probabilistic, or Monte Carlo methods. For these dynamic analyses, a series of C_e, Q_e, C_s, and Q_s values is used.

If instream toxicity measurements are available and a first order decay rate for toxicity can be estimated, the following equation should be used:

$$C = C_o e^{-K\left(\frac{x}{u}\right)}$$

where C = downstream concentration (TU_c or TU_a)
C_o = concentration after the point source discharge has mixed completely with the river (TU_c or TU_a)
x = distance downstream of complete mix point
u = velocity of river
K = measured decay rate

The DYNTOX programs, described in the subsections on dynamic modeling techniques, can be used to perform non-steady state modeling of effluent toxicity in rivers. The dilution calculation or first order decay equation can be used for continuous simulation and Monte Carlo analyses; the lognormal probabilistic method can be used for dilution calculations only.

Specific Model Recommendations — Lakes and Reservoirs

EPA's WLA guidance for modeling individual toxicants in lakes and reservoirs recommends the following steady state models [40]: SLSA, CTAP, EXAMS, and MEXAMS. As mentioned in the above section on river modeling, all of these models except MEXAMS can simulate the fate and transport of both organic chemicals and heavy metals. Only SLSA is limited to single reach and single pollutant source analyses, the other models handle multiple reaches and sources. The DYNTOX program, discussed in the dynamic modeling subsections, can be used to develop input data to run these steady state fate and transport models with Monte Carlo techniques to predict concentration-frequency curves.

The following continuous simulation models for individual toxicants in lakes and reservoirs are also recommended: TOXIWASP, CHNTRN, WASTOX, SERATRA, HSPF, and Toxic Organic Substance Transport and Bioaccumulation Model (TOXIC). All these models are designed for multiple segment, multiple pollutant source analysis of both organic chemicals and heavy metals. The DYNTOX program, discussed in the dynamic modeling subsections, can be used to develop the input data for these continuous simulation fate and transport models to predict concentration-frequency curves.

The same equations used for toxicity analyses in rivers can also be used in a steady state, continuous simulation, or probabilistic analysis of long, narrow, shallow impoundments with high inflow velocities. Wider, deeper lakes require more complicated analyses since prolonged detention times (>20 days) and stratification exert a significant impact on water quality. The prolonged detention times make it essential that receiving water measurements of toxicity be available to estimate decay factors. These measurements should be made at least once per season in order to identify any time-varying trends in toxicity fate processes. Steady state or continuous simulation fate and transport models for lakes can then be run with monitored decay rates for toxicity. Toxicity is expected to be additive even when the toxicity of one effluent affects invertebrates and the toxicity of another affects fish or algae. A simple steady state analysis can be performed using the following equations [45]:

$$T_w = V/Q$$

$$C = C_{in}/(1+T_w K)$$

where T_w = mean hydraulic residence time
V = lake volume at critical conditions
Q = mean total inflow rate at critical conditions
C = steady state lake concentration (TU_c or TU_a)
C_{in} = steady state inflow concentration (TU_c or TU_a)
K = first order decay rate

If effluent is discharged to a stratified lake and mixes only with the hypolimnion or epilimnion, the volume of that layer only should be used to calculate mean hydraulic residence time (T_w). The mean total inflow rate (Q) and the inflow concentration (C_{in}) should be calculated as the sum of all sources to the lake, including point source, nonpoint source, and tributary inputs.

Specific Model Recommendations — Estuaries and Bays

EPA's WLA guidance for modeling individual toxicants in estuaries recommends the following steady state models [41]: EXAMS and MEXAMS. EXAMS is designed for heavy metal and organic chemical analyses; MEXAMS predicts the fate of ionic species of heavy metals. Both models handle multiple segments and multiple pollutant sources. The DYNTOX program, discussed in the dynamic modeling subsections, can be used to develop input data to run these steady state fate and transport models with Monte Carlo techniques to predict concentration-frequency curves.

EPA's WLA guidance recommends the following continuous simulation models for individual toxicants in estuaries: WASTOX, TOXIWASP, and FETRA. These models are designed for multiple segment, multiple pollutant source analyses of both organic chemicals and heavy metals. The DYNTOX program, discussed in the dynamic modeling subsections, can be used to develop input data for these continuous simulation fate and transport models to predict concentration-frequency curves.

Dilution calculations for effluent toxicity discharges to an estuary are complicated by the oscillatory motion of the tides and possible stratification of the estuary. The prolonged detention times make it essential that field measurements of toxicity be available to estimate decay factors. These measurements should be made at least once per season in order to identify any time-varying trends in toxicity rate processes. Steady state or continuous simulation fate and transport models for estuaries can then be run with monitored decay rates for toxicity. Toxicity is expected to be additive even when the toxicity of one effluent affects invertebrates and the toxicity of another affects fish or algae. A simple steady state analysis can be performed using the following equations for a non-conservative pollutant entering from the river at the head of an estuary [45]:

$$B_i = \frac{r_i}{1-(1-r_i)e^{-kt}}$$

$$C_i = C_{i-1}\,\frac{f_i}{f_{i-1}}\,B_i$$

where r_i = exchange ratio for segment i as defined by modified tidal prism method
t = flushing time
f_i = fraction of freshwater in segment i
C_i = non-conservative pollutant concentration in segment i (TU_c or TU_a)
k = decay rate of pollutant

The following equations should be used for a non-conservative pollutant entering along the side of an estuary:

$$C_i = C_o \prod_{i=1...n} \left[\frac{f_i}{f_o}\,\frac{(r_i)}{1-(1-r_i)e^{-kt}} \right]$$

for segments downstream of outfall

$$C_i = C_o \prod_{i=1...n} \left[\frac{S_i}{S_o}\,\frac{(r_i)}{1-(1-r_i)e^{-kt}} \right]$$

for segments upstream of outfall

where C_i = non-conservative pollutant mean concentration in segment i (TU_c or TU_a)
C_o = conservative constituent mean concentration in segment of discharge
r_i = exchange ratio for segment i as defined by the modified tidal prism method
n = number of segments away from outfall ($n=1$ for segments adjacent to outfall, $n=2$ for segments next to these, etc.)
f_i = fraction of freshwater in segment i
f_o = fraction of freshwater in segment of discharge
S_i = salinity in segment i
S_o = salinity in segment of discharge
k = decay rate
t = flushing time

The details on how to calculate exchange ratios and flushing times for estuaries are included in Part 2 of EPA's Water Quality Assessment Manual [45]. This manual also describes how to perform these calculations for stratified estuaries using a two-dimensional box model analysis.

Allocation Procedures For Waste Loads

WLAs for water quality-based permits must be performed in accordance with EPA regulations [46,47]. Point sources, non-point sources and natural background sources all contribute to the total load of a pollutant in a waterbody. The allowable total maximum daily load (TMDL) is defined as the total such that any additional loading would produce a violation of water quality standards. Allocation of the TMDL must be made considering technical, socioeconomic, institutional, and political constraints. States have used various allocation schemes, and some require that a particular method be used. Table 5-4 shows 19 potential allocation strategies [48]. In all cases, the antidegradation provisions and the other requirements of the water quality standards regulations must be met [47].

The most commonly used allocation methods can be summarized as: 1) equal percent removal, 2) equal effluent concentrations, and 3) a hybrid method. The equal percent removal approach has two versions — the overall removal efficiencies of each pollutant source are required to be equal or the incremental removal efficiencies must be equal. The equal effluent concentration approach can also be applied in two versions — equal final concentrations or equal incremental concentration reductions. This method is similar to the equal percent removal method if influent concentrations at all sources are approximately the same. If one plant has substantially higher influent levels, however, requiring equal effluent concentrations will result in higher overall treatment levels than the equal percent removal approach.

The third commonly used method of allocating waste loads is a hybrid method in which the criteria for waste reduction may not be the same for each point source. One facility may be allowed to operate unchanged, whereas another may be required to provide the entire load reduction. More generally, a proportionality rule may be assigned that requires the percent removal to be proportional to the input loading to the treatment plant. In such situations, larger sources would be required to achieve higher overall removals.

Several different allocation schemes should be analyzed before a final strategy is selected. In selecting the

Table 5-4. Waste Load Allocation Methods

1. Equal percent removal (equal percent treatment)
2. Equal effluent concentrations
3. Equal total mass discharge per day
4. Equal mass discharge per capita per day
5. Equal reduction of raw load (pounds per day)
6. Equal ambient mean annual quality (mg/l)
7. Equal cost per pound of pollutant removed
8. Equal treatment cost per unit of production
9. Equal mass discharged per unit of raw material used
10. Equal mass discharged per unit of production

11a. Percent removal proportional to raw load per day
11b. Larger facilities to achieve higher removal rates
12. Percent removal proportional to community effective income
13a. Effluent charges (dollars per pound, etc.)
13b. Effluent charge above some load limit
14. Seasonal limits based on cost-effectiveness analysis
15. Minimum total treatment cost
16. BAT (industry) plus some level for municipal inputs
17. Divide assimilative capacity to require an "equal effort among all dischargers"
18a. Municipal: treatment level proportional to plant size
18b. Industrial: equal percent between BPT and BAT, i.e.,

$$\text{Allowable} = \text{BPT} - \frac{x}{100}(\text{BPT} - \text{BAT})$$

19. Industrial discharges given different treatment levels for different stream flows and seasons. For example, a plant might not be allowed to discharge when stream flow is below a certain value. Above that value, but below another value, the plant would be required to use a higher level of treatment than BPT. Finally, when stream flow is above an upper value, the plant would be required to treat to a level comparable to BPT.

Source: Chadderton *et al* (19).

final allocation method, the relationship between uncertainty of prediction versus cost should be examined. Uncertainty can vary between allocation schemes, as discussed by Chadderton et al. [48].

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6. *Permit Requirements*

Overview

At this point in the water quality control process, a wasteload allocation (WLA) has been developed for a discharger if violations of the State standards are predicted or occuring. The WLA is based on ambient criteria and the exposure of the resident aquatic community to toxic conditions. Where human health is of concern, the WLA is based on safe levels and the route of exposure. The next step is to derive permit limits to enforce the WLA. In certain cases permit limits will be different than the WLA values.

The WLA provides a measure of effluent quality that is necessary to protect water quality in the receiving water. It is important to consider how the WLA addresses variability in effluent quality. For example, a WLA for nutrients or bioaccumulative pollutants could be expressed as the required average effluent quality because the total loading of these pollutants is of concern. On the other hand, a WLA for toxic pollutants should be expressed as a maximum value for the effluent because the concentration of these pollutants is of more concern than the total loading. It is important to recognize that the duration and frequency with which the required effluent quality level is exceeded are important aspects of a WLA.

Permit limits are designed to require a particular level of effluent quality. Effluent quality is variable, and limits are set at a level so that if the plant maintains the desired level of performance, the probability of exceeding the limits is very low (i.e., the probability is ≤ 0.05). If limits are set too high, a discharger *not meeting* the desired level of performance would not exceed the limits when typical monitoring frequencies are employed. If limits are set too low, a discharger *meeting* the desired level of performance would often exceed the limits. In either case, determination of compliance and enforcement would be compromised.

There is a significant risk of incorrectly enforcing a WLA if effluent variability and the probability basis for both the WLA and the permit limits are not considered. For example, a steady state WLA may specify an effluent value with the assumption that it is a value never to be exceeded. The same value used as the daily maximum permit limit could allow the WLA value to be exceeded perhaps an unacceptable amount of time without observing permit violations using typical monitoring requirements. Even more confusion could result in translating a longer duration WLA requirement (e.g., a four-day average) into daily maximum and average daily permit limits. Much of this section, therefore, concerns ensuring that permit limits are derived to implement a WLA requirement correctly.

This section describes the basic principles of permit limit derivation and provides recommendations for deriving limits from various types of WLA output. This section also addresses other types of permit requirements and expression of limits.

Basic Principles of Effluent Variability

Effluent quality varies over time. If BOD data for a typical sewage treatment plant are plotted against time, the day-to-day concentration variations can be seen (see Figure 6-1a). Some of this behavior can be described by constructing a frequency-concentration plot of the same data (see Figure 6-1b). From this plot, we see that

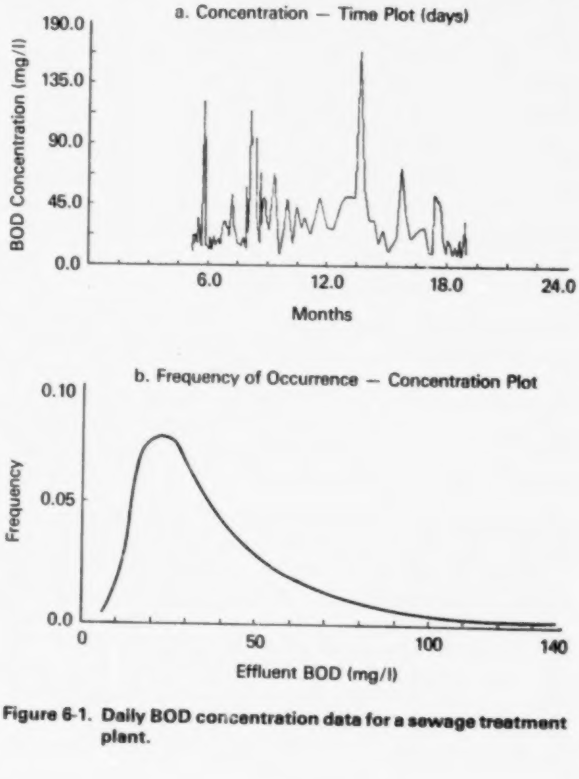


Figure 6-1. Daily BOD concentration data for a sewage treatment plant.

most of the time, BOD concentrations are near the average value. The amount of data at low concentrations is limited by the maximum treatment efficiency. There is much more data available at higher concentrations, resulting from periods when treatment efficiency drops.

From the shape of this curve, a standard statistical distribution can be fitted. The choices for statistical distributions include normal (bell shaped), lognormal (skewed), or other variations on the lognormal distribution. From the vast bulk of data that has been examined in this manner, it is reasonable to make a first assumption that treated effluent data follow a lognormal distribution.

Consequently, any treatment system can be described by using the mean concentration of the parameter of interest (i.e., the long term average [LTA]) and the variance (or coefficient of variation [CV]) and by assuming a particular statistical distribution (usually lognormal). The advantage of describing effluent characteristic behavior statistically is that the entire distribution of values can be projected from limited data, and limits can be set at a known probability of occurrence. Figure 6-2 illustrates a hypothetical distribution of daily data.

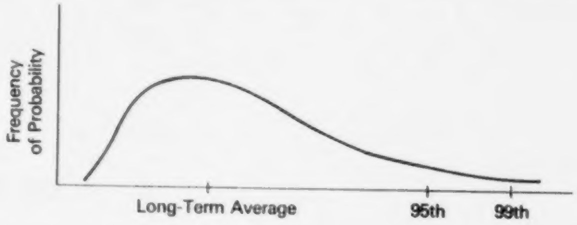


Figure 6-2. Hypothetical daily effluent parameter probability distribution.

Permit limits are set at the upper bounds of acceptable performance and are values not to be exceeded. Requirements are usually expressed using two types of permit limits. The *daily maximum* permit limit is the maximum allowable value for any single observation. The *average daily* or "monthly" permit limit is the maximum allowable value for the average of all observations obtained during one month (average daily limits for weekly periods are also used for publicly owned treatment works [POTWs]). If permit limits are set too high relative to the LTA, a discharger not complying with the desired level of performance will not exceed the limits. If permit limits are set too low, a discharger that is complying with the desired level of performance may frequently exceed the permit limits.

In setting the average daily (i.e., monthly) limit, the number of samples used to compute the average is a factor. As the number of observations (n) increases, the probability distribution of the n-day average values becomes narrower around the LTA. Therefore, the 95th or 99th percentile value for an n-day average will approach the LTA as n increases.

The frequency of monitoring should determine the value of n for calculating the average daily (monthly) limit. However, if limits are derived using a value for n that is too low, the limit will be high and a discharger could disguise a poorly operating plant by increasing sampling during periods of good operation to lower the reported average. Therefore, some intermediate basis may be needed for average daily limits regardless of actual monitoring frequency. EPA recommends basing n on the actual monitoring frequency where it is expected that monitoring frequency will not exceed the required frequency. For limits that are inexpensive to monitor (such as metals) n should not be less than 10. EPA has used $n = 10$ as the basis for average daily requirements in many effluent limitations guidelines.

Permit Limit Derivation

Limits for any parameter can be derived for any plant from the LTA and CV for the parameter. In many cases, the LTA and CV values can be estimated. The same procedures are used for technology-based and water quality-based requirements. The actual pollution control requirements are dictated by the LTA and CV. Table 6-1 presents the general procedures for deriving daily maximum limits and average daily limits. Technical statistical details of these procedures may be found in Appendix E and References 1 and 2.

The procedures in Table 6-1 for calculating the average daily (monthly) limits are based on an assumption that monitoring observations are independent of (uncorrelated with) each other. This is a reasonable assumption for most industrial treatment plants. Some treatment plants (particularly large biological systems) exhibit serial correlation of monitoring observations. In such cases it would be more accurate to use an estimate of the variance of the n-day averages that incorporates this correlation. Such an estimate of the CV can be generated from historical data where treatment will not change significantly. For example, to derive an estimate for $n = 8$, construct a data set of eight-day averages that approximates the monitoring that will be used for the monthly limit (e.g., every Tuesday and Friday). Then derive the CV for those values. Calculation of the average daily permit limit in this case would then be identical to the procedures used to calculate the daily maximum in Table 6-1 except that the CV for the n-day averages would be used instead of the CV for daily values.

Ensuring Consistency with the WLA

As previously stated, the WLA for a discharger will already have been completed. The permit writer should be especially careful to ensure that the limits designed to enforce the WLA are consistent with the assump-

Table 6-1. Statistical derivation of permit limits.

Definition of Statistics

$X_1, X_2, \ldots, X_k$ daily pollutant measurements

$E(X)$ = long-term average = $(X_1 + X_2 + \ldots + X_k)/k$

$V(X)$ = variance = $\sum_{i=1}^{k} (X_i - \overline{X})^2/(k-1)$

CV = coefficient of variation = $\sqrt{V(X)/E(X)^2}$

Derivation of Limits

Given values for E(X) and CV proceed as follows:

1. Estimate σ^2 as $\sigma^2 = \ln(CV^2 + 1)$
2. Estimate μ as $\mu = \ln E(X) - .5\sigma^2$
3. Calculate daily limits as $e^{\mu} + Z\sigma$
   a. For 95th percentile limit $Z = 1.645$, therefore, daily limit$_{95} = e^{\mu} + 1.645\sigma$
   b. For 99th percentile limit $Z = 2.326$, therefore, daily limit$_{99} = e^{\mu} + 2.326\sigma$
4. Calculate $\sigma_n^2 = \ln[1 + ((e^{\sigma^2} - 1)/n)]$
   where n = assumed observations per month
5. Calculate $\mu_n = \mu + (\sigma^2 - \sigma_n^2)/2$
6. Calculate average daily (monthly) limit as $e^{(\mu_n + Z\sigma_n)}$
   a. For 95th percentile limit $Z = 1.645$, therefore, average daily limit$_{95} = e^{\mu_n} + 1.645\sigma_n$
   b. For 99th percentile limit $Z = 2.326$, therefore, average daily limit$_{99} = e^{\mu_n} + 2.326\sigma_n$

Example

Assume E(X) = 4 ug/l and CV = 1.0 and interested in 10-day average (n = 10)

1. $\sigma^2 = \ln(1.0^2 + 1) = .6931$
2. $\mu = \ln 4 - .5\,(.6931) = 1.0397$

3a. Daily limit$_{95}$ = $\exp[1.0397 + 1.645\sqrt{.6931}]$ = 11.1 ug/l

3b. Daily limit$_{99}$ = $\exp[1.0397 + 2.326\sqrt{.6931}]$ = 19.6 ug/l

4. $\sigma^2_{10} = 1n\,[1 + ((e^{.6931} - 1)/10)] = .0953$
5. $\mu_{10} = 1.0397 + [(.6931 - .0953)/2] = 1.3386$

6a. Average daily limit$_{95}$ = $\exp[1.3386 + 1.645\sqrt{.0953}]$ = 6.3 ug/l

6b. Average daily limit$_{99}$ = $\exp[1.3386 + 2.326\sqrt{.0953}]$ = 7.8 ug/l

Derivation of Average Daily Limit Based on 30 Samples

Given values for E(X) and CV proceed as follows:

For 30-day average limit$_{95}$ calculate $E(X)\,[1 + \frac{1.645}{\sqrt{30}}(CV)]$

For 30-day average limit$_{99}$ calculate $E(X)\,[1 + \frac{2.326}{\sqrt{30}}(CV)]$

Example

Assume E(X) = 4 ug/l and CV = 1.0

30-day average limit$_{95}$ = $4\,[1 + \frac{1.645}{\sqrt{30}}(1.0)]$ = 5.2 ug/l

30-day average limit$_{99}$ = $4\,[1 + \frac{2.326}{\sqrt{30}}(1.0)]$ = 5.6 ug/l

tions and requirements associated with the WLA. Potential problem areas are described below.

- A WLA requirement may be based on a particular probability that the WLA limit will be exceeded. This probability may not be consistent with the probability used to set limits.
- A WLA that assumes the discharge is steady state (i.e., not changing over time) requires an assumption regarding how the effluent may vary.
- Daily maximum and average daily (monthly) limits must be developed so that they are consistent with each other and reflect the required level of performance.
- A WLA value specified as an average (e.g., a seven-consecutive-day average) must be translated into daily maximum and average daily limits.

There are several ways that a WLA may be developed and expressed. Table 6-2 presents the major types of

WLA output and options for deriving limits. Each of the WLA output types is addressed below.

WLA Output Type 1 — Required Effluent Performance

The least ambiguous way that a WLA can be specified is as the required effluent performance in terms of the LTA and CV of the daily values. When a WLA is expressed as such, there is no confusion about assumptions used and the translation to permit limits. A permit engineer can easily design permit limitations to achieve the WLA objectives.

The types of exposure analyses that yield a WLA in terms of required performance are the continuous simulation, Monte Carlo, and lognormal probabilistic analyses. Although these analyses are slightly more complex than a steady state analysis, they are preferred because they specifically address effluent variability. Interactive computer programs that are available to facilitate use of these WLA models are described in Section 5.

The permit limit derivation procedures described above are used to derive daily maximum and average daily (monthly) limits from the required effluent LTA and CV. As described above, EPA generally uses 10-day averages to derive the daily average limit (regardless of actual monitoring frequency). Depending on the monitoring that is expected to be done, other values of n may be used in deriving the daily average limits on a case-by-case basis.

WLA Output Type 2 — Single Value from a Steady State Analysis

The vast majority of WLAs are reported as a single value for effluent quality. An example of such a requirement is "copper concentration must not exceed 0.75 milligrams per liter (mg/l)." Steady state analyses assume that the effluent is constant and, therefore, that the WLA value will never be exceeded. This presents a real problem in deriving permit limits because permit limits must reflect variability. A second problem is that the effluent concentrations may be occasionally greater than the permit limits without registering limit violations because of the frequency of the monitoring used.

The proper enforcement of this type of WLA depends on the parameter limited. For nutrients, BOD, and bioaccumulative pollutants, the WLA value should generally be used as the average daily permit limit. However, the impact associated with toxic pollutants is much more time dependent as reflected in the four-day average duration for the CCC (see Section 2). Therefore, the WLA value for toxic pollutants should be used as the daily maximum permit limit.

The disadvantage of this approach is that the average daily limits must be derived without any information about the variability of the effluent parameter. Without other information, most permit engineers divide the daily maximum limit by 1.5 or 2.0 to derive an average daily limit (depending on the expected range of variability).

The second approach available when a WLA is derived as a single value using a steady state assumption is to use the methodology described in Reference 3. This methodology is used to determine whether it is appropriate to enforce the WLA result on a basis less stringent than the daily limit. The methodology involves redoing the WLA using the lognormal probability analysis to determine the frequency with which the CMC (the one-day criterion) would be exceeded under three separate scenarios for permit limits.

WLA Output Type 3 — Steady State Values with a Specified Duration or with a Specified Permit Limit Probability Basis

WLAs may begin to be reported as four- or seven-day average values that then would require translation to permit limits. The "toxicologically based steady state" WLA approach provides one-day and four-to-seven day average requirements based on acute and chronic protection. Also, if a WLA value did have a specified probability of being exceeded, permit limits could then be derived to enforce the requirement consistent with the probabilities commonly used to set limits. The following methodology was developed to accomplish these objectives. The methodology consists of the following general steps:

- Step 1 — An effluent performance level (LTA and CV) that will meet the WLA requirement is back-calculated. Where two requirements are specified based on different duration periods, two performance levels are back-calculated.
- Step 2 — Permit limits are then derived directly from whichever performance level is more restrictive.

The benefits of this methodology are: 1) limits may be adjusted relative to the fixed value WLA to account for a different permit limit probability basis, 2) the daily maximum and average daily limits can be made consistent to maximize enforceability, and 3) WLA requirements specified as an average can be translated to standard permit limits.

To adjust limits to correct for probability basis requires that the probability basis for the WLA value must be specified. Steady state WLAs usually mean the value may *never* be exceeded. If "never" was interpreted to mean no more than one day per year, for example, the probability would be 0.003 (1 day/365 days). Making an assumption about the probability that the WLA may be exceeded allows this methodology to be used to correct for inconsistent probability bases.

Table 6-2. Wasteload allocation (WLA) outputs and derivation of limits.

WLA Output Type	Derivation of Limits
1. Required effluent performance	Derive maximum and average limits directly
2. Single value from a steady state analysis	Incorporate value as daily maximum limit Use reference [3] to assess variability and the degree to which the value can be incorporated as an average daily limit
3. Steady state values with specified duration or specified exceedence probability	Derive required long-term average/variability estimates for each requirement; select limiting requirement; derive limits from long-term average/variability

Table 6-3. Typical coefficients of variation

Sewage Treatment[1]

Treatment Process	Number of Plants	Coefficient of Variation BOD	Coefficient of Variation TSS
Trickling filter rock	64	.40	.50
Trickling filter plastic	17	.50	.65
Conv. act. sludge	66	.65	.86
Contact stabilization	57	.60	.70
Extended aeration	28	.70	.65
Rotating bio. contactors	27	.60	.70
Oxidation ditch	28	.60	.70
Stabilization pond	37	.50	.65

Chemical Precipitation/Settling[2]

Pollutant	Coefficient of Variation
Cr	.99
Cu	.60
Fe	.57
Mn	.84
Ni	.81
Zn	.84
Tss	.66

Pharmaceutical Industry[3]

Plant Number	BOD	(n)	TSS	(n)
	Coefficient of Variation			
12015	1.01	46	.85	195
12072	.97	392	.63	395
12026	.95	44	.49	53
12036	.74	366	1.12	364
12097	1.08	222	1.21	249
12098	1.37	24	1.52	25
12117	.70	39	.81	51
12160	.92	34	1.11	32
12161	.56	249	.99	365
12186	.71	54	.50	54
12187	.21	12	.26	12
12136	1.02	110	1.16	111
12248	.58	50	.55	52
12257	.64	56	.92	56
12294	.93	56	1.25	50
12307	1.56	39	1.34	38

[1] From Table C-2, Reference [1].

[2] From Table 3, page 14 of 10-18-83 memorandum from H. Kahn to E. Hall titled, "Revisions to Data and Analysis of the Combined Metals Data Base."

[3] From preliminary descriptive statistics generated on pharmaceutical data by SRI International, 11-12-82.

To use the methodology only to translate a WLA specified as an average to permit limits (and not correct for probability), merely use the same probability for the WLA as that used for deriving limits (e.g., 0.05). The methodology is described below in five steps.

Step 1 — Estimate Effluent Variability

The first step is to estimate the daily CV the parameter will have after the plant complies with the requirements. If variability is mostly related to production, current data may be used. If future variability will be substantially different, the CV must be estimated. Table 6-3 provides data on effluent parameter CVs. (The variability estimates for sewage treatment reflect secondary treatment, not advanced secondary treatment.) Discharges of toxic pollutants are generally more variable than discharges of conventional pollutants. It is important to use an estimate of the CV that is as low as can be reasonably achieved. In this procedure, the LTA is back-calculated using the CV and the WLA requirement. Table 6-4 illustrates how the CV affects the relationship between the LTA and various measures of extreme performance.

Step 2 — Derive Performance for One-day WLA Requirement

The second step is to determine an effluent distribution that will meet the one-day WLA requirement (here assumed to be acute protection). Toxicity requirements must be converted to toxic units (TUs) (100/toxicity in % effluent) in order to use these equations. The WLA will have been conducted assuming some probability of occurrence of the WLA value. Where the WLA does not specify an assumed effluent occurrence frequency, use a required occurrence probability of 0.01 or 0.05 (i.e., the 99th or 95th percentile). As explained above, the assumed probability that the WLA will be exceeded is not necessarily the same as the probability basis for permit limit calculation. One purpose of this methodology is to derive different values based on different probability bases for the WLA and permit limits.

Using the acute protection effluent value (WLA), the required probability of occurrence, and the CV, derive the log mean using:

$$\mu = \ln\ (\text{acute WLA}) - Z\sigma$$

where: acute WLA = the effluent acute protection value, Z = the critical value statistic (1.645 for 0.05 probability and 2.326 for 0.01 probability), and $\sigma = \sqrt{\ln(CV^2 + 1)}$.

The LTA is derived from μ and σ using:

$$\text{LTA} = e^{\mu + .5\sigma^2}$$

Table 6-4. Relationship between coefficient of variation and various permit limits for the same long-term average.

Long-Term Average (ppb)	Coefficient of Variation	Permit Limits (ppb)					
		Daily Maximum		Daily Avg. (10-day)		Daily Avg. (30-day)	
		95%	99%	95%	99%	95%	99%
4	0.5	7.8	10.7	5.1	5.7	4.6	4.8
4	1.0	11.1	19.6	6.3	7.8	5.2	5.6
4	2.0	14.4	34.2	8.8	13.0	6.4	7.4

Long-Term Average (LC_{50} in % effluent)	Coefficient of Variation	Permit Limits (LC_{50} in Percent Effluent)					
		Daily Maximum		Daily Avg. (10-day)		Daily Avg. (30-day)	
		95%	99%	95%	99%	95%	99%
25	0.5	12.8	9.3	19.6	17.5	21.7	20.8
25	1.0	9.0	5.1	15.8	12.8	19.3	17.9
25	2.0	6.9	2.9	11.4	7.7	15.6	13.5

Step 3 — Derive Performance for Average WLA Requirement

The third step is to derive an effluent distribution that will meet the average WLA requirement (here assumed to be derived from the CCC). For example, the WLA value may be specified as a four-day average. As with the one-day WLA, unless the WLA specifies a required probability of occurrence for the average WLA value, use a required occurrence probability of 0.01 or 0.05.

The object is to derive a required distribution of averages that will meet the WLA. From this distribution of averages, a distribution of daily values is derived because permit limits will later be calculated from whichever distribution of daily values is limiting. The equations below assume independence of daily effluent values (i.e., one day's value is independent of the previous day's). This assumption is accurate for the majority of industrial sources. Effluent data from sewage treatment plants and industrial biological systems, however, are more likely to exhibit serial correlation. To better reflect such sources, the required distribution of average values can be calculated using a variance estimate that reflects the serial correlation of data. The procedures for calculating the required LTA where serial correlation is strong are explained in Appendix E.

For illustration, the equations below assume the WLA specified a seven-day average. If the WLA specifies a different duration, substitute that value for "7" in the equations. Use the chronic protection effluent value (WLA), the required probability of occurrence, and the CV to derive the log mean of the average values using:

$$\mu_7 = \ln\ (\text{chronic WLA}) - Z \sqrt{\ \ln\ [1+((e^{\sigma^2}-1)/7)]}$$

where μ_7 = the log mean of the seven-day averages, chronic WLA = the chronic protection value, and σ and Z are as described in Step 2. Next derive the log mean of daily values using:

$$\mu = \mu_7 - .5\sigma^2 + .5\ \ln\ [1+((e^{\sigma^2}-1)/7)]$$

Finally, derive the LTA of the daily values using:

$$\text{LTA} = e^{\mu+.5\sigma^2}$$

Step 4 — Evaluate the Required Performance

Compare the two LTAs from Steps 2 and 3. The *lower* average indicates which WLA requirement (the one-day or the average) is more limiting for the case in question. This average provides the required effluent characteristic distribution in terms of an *LTA and daily CV.* Examine the distribution from an engineering viewpoint. Is the LTA feasible for the plant? Does the variance estimate still hold? For example, if meeting the distribution would require substantially better process control, you might consider reestimating the CV based on a different treatment system and recalculating the means (Steps 2 and 3).

Step 5 — Derive Permit Limits

The last step is to calculate daily maximum and average daily limits to reflect the effluent performance that meets the WLA. The procedures described above under the permit limit derivation section are used.

Expressing Limitations

For limitations on specific chemicals, the daily maximum and average daily or "monthly" limits as calculated above may be used directly in Part I of the permit. If the monitoring frequency is once per month, the daily maximum and average daily limits should be the same. Section 7 describes options involving variable frequency of monitoring or compliance demonstration followed by reduction of monitoring frequency. For example, the first year of the permit might require twice-weekly monitoring and contain limits expressed as daily maximum and average daily (monthly). After one year, the frequency could be dropped to some lesser frequency and the average daily limit altered if compliance had been demonstrated. Such a change, if it had been specified in the original permit, would not require modification of the permit.

The average daily limit is related to the monitoring frequency required, but the LTA and the CV of the effluent concentrations do not change. Water quality-based treatment facilities should be designed and built to achieve a performance level consistent with the WLA or the required LTA and CV. Sections 124.8 and 124.56 of 40 CFR Part 124 require a fact sheet containing "any calculations or other necessary explanation of the derivation of specific effluent limitations . . ." for every draft permit. Accordingly, the WLA along with the required LTA and CV, where available, must be included in the fact sheet. The fact sheet should also state that the required LTA and CV rather than the permit limit should be used for the design of the treatment facility.

When toxicity limits are used, additional description of the limit is required. The limit should be stated in Part I as "effluent toxicity" in the parameter column with "maximum TUs," "minimum LC50," or "minimum NOEL" in parenthesis underneath. The numerical values should be placed in the appropriate concentration column followed by TU or a percent sign. A footnote should direct the reader to Part III, Additional Conditions. If the monitoring requirements cannot be fitted into the columns in Part I, a reference such as "see Part III" should be used.

The further description in Part III should accomplish the following:

- Explain how the limit is expressed (e.g., the limitation is the minimum LC50 expressed as percent effluent, or the limitation is the maximum acute toxic units [TU_a]).
- Specify the test organism and the test methods.
- Describe any special reporting or followup requirements (e.g., requirements to conduct a toxicity reduction evaluation).

The language in Part III should be modified as needed to suit the situation. The following language should be used only as an example:

- The effluent toxicity limitation contained in Part I is the allowable chronic toxicity to the most sensitive of three test species. It is expressed as the maximum NOEL in percent effluent. The required test species and the procedures to follow are described in Methods for Measuring the Chronic Toxicity of Effluents to Aquatic Organisms, EPA-600/4-85-014, August, 1985.
- The permittee shall conduct monitoring of effluent toxicity once per month. One 24-hour composite sample shall be collected and tested within 24 hours of collection. Results shall be reported as the NOEL. Any test that does not meet quality control requirements as described in the above referenced methods shall be repeated using a freshly collected sample as soon as practicable.
- If the permittee does not comply with the effluent toxicity limitation in Part I, the permittee shall submit, if requested by the Director, within 45 days, a plan and schedule for conducting a toxicity reduction evaluation. The toxicity reduction evaluation, when completed, shall determine how the permittee can achieve the effluent toxicity limitation including an implementation schedule. After review of the plan by EPA, the permittee shall conduct the evaluation within the specified time frames. Upon completion of the toxicity reduction evaluation, this permit may be modified, or alternatively revoked and reissued, in order to incorporate appropriate permit conditions and compliance schedules.

Toxicity Reduction Evaluation and Indicator Limits

One mechanism that can be used in bringing a discharger into compliance with a difficult water quality-based requirement is a toxicity reduction evaluation (TRE). A TRE is a study conducted to determine what control options are effective for complying with either toxicity or chemical concentration requirements. In most cases the plant manager should be responsible for conducting a TRE. A TRE can be done prior to permit issuance, during the permit term in response to a monitoring trigger, during the permit term in response to limits being exceeded, or in response to an administrative order.

The purpose of a TRE is threefold: 1) to isolate causative pollutants or manufacturing processes that produce the chemicals of interest; 2) to identify control options and determine the effectiveness of each option; and 3) to identify a compliance monitoring indicator and demonstrate its effectiveness. Of these three purposes, only the second is essential to developing a control plan for the facility. Toxicity may be used as a control parameter without identifying causitive chemicals.

The first step of a TRE is to try to determine what chemicals are causing toxicity or, for specific chemicals, what manufacturing processes are producing the pollutants. This can be accomplished by an analysis of feedstocks, products, by-products, and treatment design and operation. Chemical analysis, in-plant toxicity testing, and wastewater chemical fractionization followed by toxicity or chemical analysis of the fractions may also be useful to isolate causes. A fractionation procedure to isolate toxic constituents and fractions has been developed [4]. The principal advantage to isolating causative factors is that cost-effective process modifications may be available.

The second and essential step is to determine how effective alternative control options are in reducing the discharge of toxic pollutants. If specific chemicals are involved, the effects of process modifications or treatment changes can be estimated using chemical and treatability data, or determined through bench-scale studies. When effluent toxicity is involved, bench-scale studies should be used to determine the reduction in wastewater toxicity associated with different control options. The draft 301(g) guidance contains information on treatability studies [5]. EPA is also developing industry-specific studies of toxicity control [6-10].

The third step is to explore the use of an indicator limit. An indicator limit is a limit on some pollutant or parameter that will ensure compliance with a separate chemical or toxicity requirement. The burden of proof for establishing indicator limits rests with the discharger. The benefits of using an indicator for compliance monitoring are lesser cost for the permittee and greater frequency of monitoring and consequently better enforcement for the regulatory agency.

The utility of an indicator limit must be demonstrated by the permittee. Such a demonstration may be based on: 1) similarities between causative pollutants and a convenient monitoring parameter (e.g., oil and grease limits to ensure control of polynuclear aromatic hydrocarbons) or 2) the effectiveness of a particular treatment process and a control parameter for that process (e.g. limits on total organic carbon [TOC] to ensure proper performance of an activated carbon process). Where an indicator limit is used, periodic monitoring of toxicity to test the validity of the indicator limit should be required.

Other Permit Requirements

Operational Controls

The most effective control of toxic pollutant discharges may result from operational requirements. For example, a TRE or other sources may reveal that spills during material handling are causing discharges of toxic chemicals. Requirements for spill reduction, containment, and treatment may eliminate the discharge. Such requirements may be instituted as Best Management Practices (BMP) requirements under sections 402(a) and 304(e) of the Clean Water Act (see 40 CFR Part 125 Subpart K). Additional technical information is contained in a publication entitled NPDES Best Management Practices Guidance Document [11].

Anti-degradation and Anti-backsliding

The procedures in this document for developing water quality-based permit limits for toxics at a source will normally result in new or more stringent water quality-based limits than those contained in a previously issued permit. In a limited number of cases, however, it is conceivable that less stringent water quality-based limits could result. In these cases, permit limits must conform to existing Federal regulations governing both anti-degradation (existing instream water uses shall be maintained and protected) and anti-backsliding (issuance of permit limits that are less stringent than those contained in the existing permit is prohibited). The pertinent regulations are 40 CFR Part 131 (see especially §131.12) and 40 CFR Part 122 (see especially §122.44(l) and 122.62). Anti-backsliding is frequently the subject of debate in State legislatures and in Congress. Permit writers should keep apprised of recent statutory or regulatory developments in this area.

Data Collection

Section 308 of the Clean Water Act and corresponding State statutes authorize the imposition of monitoring and data collection requirements on any owner or operator of a point source discharge. The only limitation on this authority is a test of reasonableness. There must be a reasonable need for the information, and the schedule and costs of the requirements must be reasonable. Regulatory agencies have been granted broad discretion in defining what is reasonable.

Requirements to conduct biological assessments, toxicity reduction evaluations, and in-plant monitoring are all authorized. In a recent challenge to certain aspects of 308 authority, the court firmly upheld regulatory agencies' authority to require in-plant testing of wastestreams and manufacturing processes (see Mobil Oil Corporation vs. EPA, 716F.2d 1187, 7th Cir. 1983).

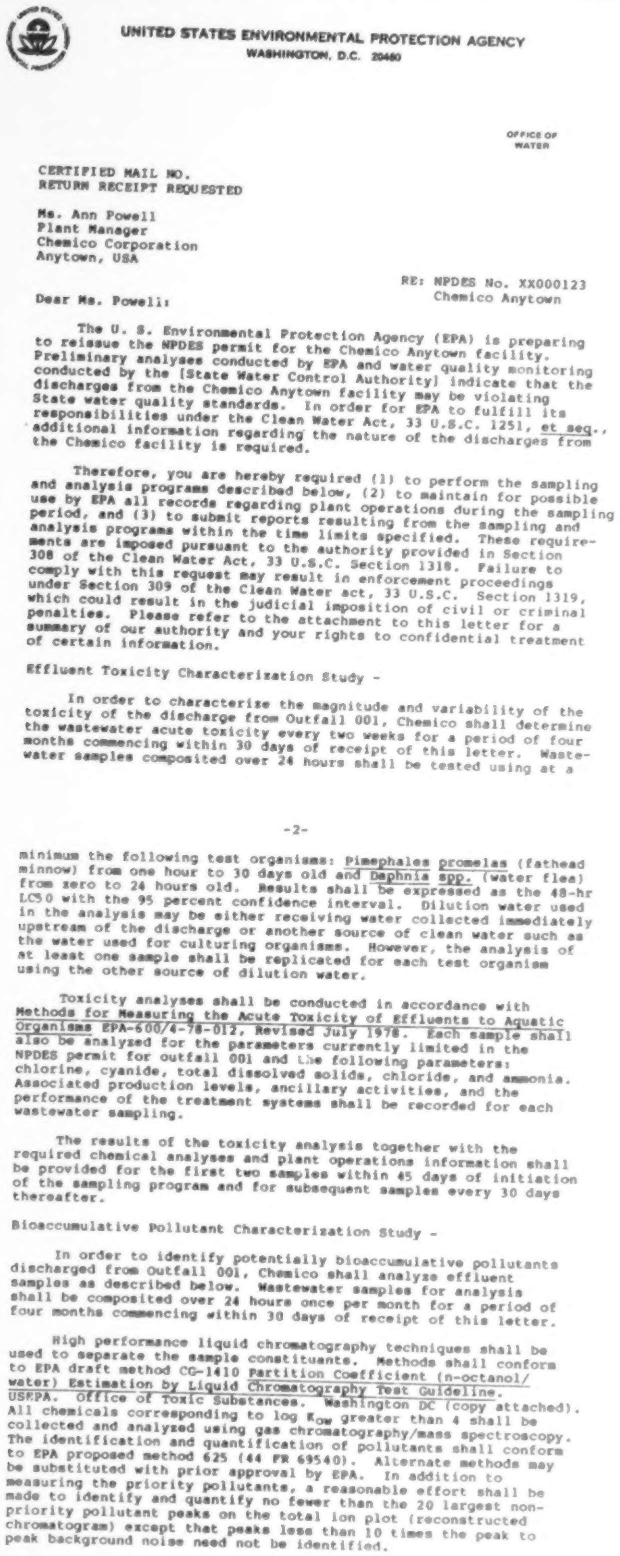
Requirements to collect data may be implemented through a direct request (commonly referred to as a "308 letter"), permit reporting requirements, or an administrative order. If sufficient data can be collected prior to permit issuance, a 308 letter is the best means to obtain the data. An example 308 letter is presented as Figure 6-3.

Permit requirements to collect data should be used when longer term data (e.g., for several seasons) are needed and there are insufficient data to develop and include water quality-based limitations in the newly issued permit. The permit should include a statement that the permit may be modified or revoked and reissued if the data indicate violation of State water quality standards. Several agencies have experienced problems in negotiating study plans based on a generalized permit requirement. If the permit requirement is non-specific (such as requiring the development and execution of a plan of study), minimum requirements should be included.

References

1. U.S. Environmental Protection Agency. 1979. Development Document for Existing Source Pretreatment Standards for the Electroplating Point Source Category, Appendix XII. EPA 440/1-79/003.
2. U.S. Environmental Protection Agency. 1982. A Statistical Analysis of the Combined Metals Industries Effluent Data, Appendix I. Report submitted by JRB Associates under EPA Contract 68-01-6347. Office of Water Regulations and Standards (WH-586), Washington, D.C.
3. U.S. Environmental Protection Agency. 1984. Technical Guidance Manual for Performing Wasteload Allocations, Book VII, Permit Averaging Periods. Office of Water Regulations and Standards (WH-553), Washington, D.C.
4. Walsh, G. E., and R.L. Garnes. 1983. Determination of Bioavailability of Chemical Fractions of Liquid Wastes Using Freshwater and Saltwater Algae and Crustaceans. Env. Sci. Tech., Vol. 17, pp. 180-182.
5. U.S. Environmental Protection Agency. Draft Technical Guidance Manual for Regulations Promulgated Pursuant to Section 301(g) of the Clean Water Act. Appendix C—Bench Scale Treatability. Available from Permits Division (EN-336), Washington, D.C.
6. U.S. Environmental Protection Agency. 1983. Toxicity of Leather Tanning and Finishing Wastewaters. Office of Water Enforcement and Permits (EN-336), Washington, D.C.
7. U.S. Environmental Protection Agency. 1983. Toxicity Reduction Manual for the Textiles Manufacturing Industry. Office of Water Enforcement and Permits (EN-336), Washington, D.C.
8. U.S. Environmental Protection Agency. (In press). Toxicity of Iron and Steel Wastewaters. IERL, Research Triangle Park, NC.
9. U.S. Environmental Protection Agency. (In press). Toxicity of Organic Chemical Industry Wastewaters. IERL, Cincinnati, OH.
10. U.S. Environmental Protection Agency. (In press). Toxicity Reduction of Pulp and Paper Mill Wastewater. IERL, Cincinnati, OH.
11. U.S. Environmental Protection Agency. 1983. NPDES Best Management Practices Guidance Document. Office of Water Enforcement and Permits (EN-336), Washington, D.C.

Figure 6-3.
Example 308 letter.



UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF
WATER

CERTIFIED MAIL NO.
RETURN RECEIPT REQUESTED

Ms. Ann Powell
Plant Manager
Chemico Corporation
Anytown, USA

RE: NPDES No. XX000123
Chemico Anytown

Dear Ms. Powell:

The U. S. Environmental Protection Agency (EPA) is preparing to reissue the NPDES permit for the Chemico Anytown facility. Preliminary analyses conducted by EPA and water quality monitoring conducted by the [State Water Control Authority] indicate that the discharges from the Chemico Anytown facility may be violating State water quality standards. In order for EPA to fulfill its responsibilities under the Clean Water Act, 33 U.S.C. 1251, *et seq.*, additional information regarding the nature of the discharges from the Chemico facility is required.

Therefore, you are hereby required (1) to perform the sampling and analysis programs described below, (2) to maintain for possible use by EPA all records regarding plant operations during the sampling period, and (3) to submit reports resulting from the sampling and analysis programs within the time limits specified. These requirements are imposed pursuant to the authority provided in Section 308 of the Clean Water Act, 33 U.S.C. Section 1318. Failure to comply with this request may result in enforcement proceedings under Section 309 of the Clean Water act, 33 U.S.C. Section 1319, which could result in the judicial imposition of civil or criminal penalties. Please refer to the attachment to this letter for a summary of our authority and your rights to confidential treatment of certain information.

Effluent Toxicity Characterization Study -

In order to characterize the magnitude and variability of the toxicity of the discharge from Outfall 001, Chemico shall determine the wastewater acute toxicity every two weeks for a period of four months commencing within 30 days of receipt of this letter. Wastewater samples composited over 24 hours shall be tested using at a

-2-

minimum the following test organisms: *Pimephales promelas* (fathead minnow) from one hour to 30 days old and *Daphnia spp.* (water flea) from zero to 24 hours old. Results shall be expressed as the 48-hr LC50 with the 95 percent confidence interval. Dilution water used in the analysis may be either receiving water collected immediately upstream of the discharge or another source of clean water such as the water used for culturing organisms. However, the analysis of at least one sample shall be replicated for each test organism using the other source of dilution water.

Toxicity analyses shall be conducted in accordance with *Methods for Measuring the Acute Toxicity of Effluents to Aquatic Organisms* EPA-600/4-78-012, Revised July 1978. Each sample shall also be analyzed for the parameters currently limited in the NPDES permit for outfall 001 and the following parameters: chlorine, cyanide, total dissolved solids, chloride, and ammonia. Associated production levels, ancillary activities, and the performance of the treatment systems shall be recorded for each wastewater sampling.

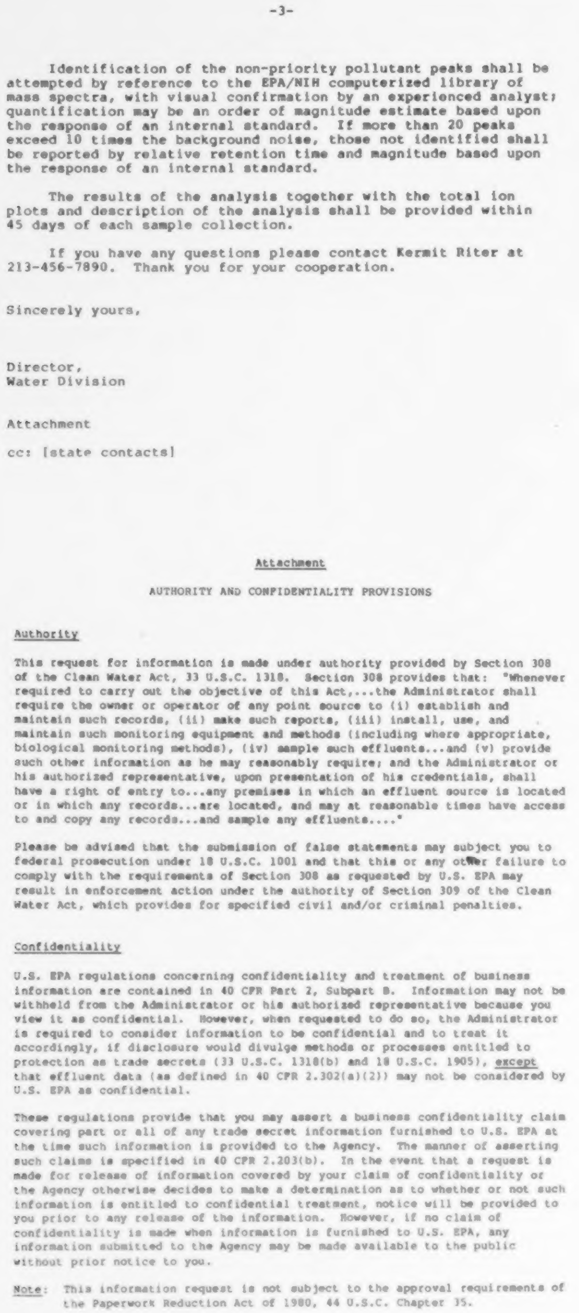
The results of the toxicity analysis together with the required chemical analyses and plant operations information shall be provided for the first two samples within 45 days of initiation of the sampling program and for subsequent samples every 30 days thereafter.

Bioaccumulative Pollutant Characterization Study -

In order to identify potentially bioaccumulative pollutants discharged from Outfall 001, Chemico shall analyze effluent samples as described below. Wastewater samples for analysis shall be composited over 24 hours once per month for a period of four months commencing within 30 days of receipt of this letter.

High performance liquid chromatography techniques shall be used to separate the sample constituents. Methods shall conform to EPA draft method CG-1410 *Partition Coefficient (n-octanol/water) Estimation by Liquid Chromatography Test Guideline*. USEPA. Office of Toxic Substances. Washington DC (copy attached). All chemicals corresponding to log K_{ow} greater than 4 shall be collected and analyzed using gas chromatography/mass spectroscopy. The identification and quantification of pollutants shall conform to EPA proposed method 625 (44 FR 69540). Alternate methods may be substituted with prior approval by EPA. In addition to measuring the priority pollutants, a reasonable effort shall be made to identify and quantify no fewer than the 20 largest non-priority pollutant peaks on the total ion plot (reconstructed chromatogram) except that peaks less than 10 times the peak to peak background noise need not be identified.

Figure 6-3 (cont.)



-3-

Identification of the non-priority pollutant peaks shall be attempted by reference to the EPA/NIH computerized library of mass spectra, with visual confirmation by an experienced analyst; quantification may be an order of magnitude estimate based upon the response of an internal standard. If more than 20 peaks exceed 10 times the background noise, those not identified shall be reported by relative retention time and magnitude based upon the response of an internal standard.

The results of the analysis together with the total ion plots and description of the analysis shall be provided within 45 days of each sample collection.

If you have any questions please contact Kermit Riter at 213-456-7890. Thank you for your cooperation.

Sincerely yours,

Director,
Water Division

Attachment

cc: [state contacts]

Attachment

AUTHORITY AND CONFIDENTIALITY PROVISIONS

Authority

This request for information is made under authority provided by Section 308 of the Clean Water Act, 33 U.S.C. 1318. Section 308 provides that: "Whenever required to carry out the objective of this Act,...the Administrator shall require the owner or operator of any point source to (i) establish and maintain such records, (ii) make such reports, (iii) install, use, and maintain such monitoring equipment and methods (including where appropriate, biological monitoring methods), (iv) sample such effluents...and (v) provide such other information as he may reasonably require; and the Administrator or his authorized representative, upon presentation of his credentials, shall have a right of entry to...any premises in which an effluent source is located or in which any records...are located, and may at reasonable times have access to and copy any records...and sample any effluents...."

Please be advised that the submission of false statements may subject you to federal prosecution under 18 U.S.C. 1001 and that this or any other failure to comply with the requirements of Section 308 as requested by U.S. EPA may result in enforcement action under the authority of Section 309 of the Clean Water Act, which provides for specified civil and/or criminal penalties.

Confidentiality

U.S. EPA regulations concerning confidentiality and treatment of business information are contained in 40 CFR Part 2, Subpart B. Information may not be withheld from the Administrator or his authorized representative because you view it as confidential. However, when requested to do so, the Administrator is required to consider information to be confidential and to treat it accordingly, if disclosure would divulge methods or processes entitled to protection as trade secrets (33 U.S.C. 1318(b) and 18 U.S.C. 1905), except that effluent data (as defined in 40 CFR 2.302(a)(2)) may not be considered by U.S. EPA as confidential.

These regulations provide that you may assert a business confidentiality claim covering part or all of any trade secret information furnished to U.S. EPA at the time such information is provided to the Agency. The manner of asserting such claims is specified in 40 CFR 2.203(b). In the event that a request is made for release of information covered by your claim of confidentiality or the Agency otherwise decides to make a determination as to whether or not such information is entitled to confidential treatment, notice will be provided to you prior to any release of the information. However, if no claim of confidentiality is made when information is furnished to U.S. EPA, any information submitted to the Agency may be made available to the public without prior notice to you.

Note: This information request is not subject to the approval requirements of the Paperwork Reduction Act of 1980, 44 U.S.C. Chapter 35.

7. *Compliance Monitoring*

Overview

Section 308 of the Clean Water Act and equivalent State statutes authorize the imposition of monitoring requirements "whenever required to carry out the objectives of this Act" and require the permittee to grant access to treatment facilities and records for inspections by regulatory authorities and their designated agents. Whenever the permit limits the value of a discharge parameter, the permit will also require self-monitoring of that parameter. The permit may also include monitoring of parameters that are not limited. Regulatory authorities conduct inspections of permitted facilities to verify compliance status.

The permittee is responsible for conducting monitoring, maintaining records, and submitting reports. In addition, the permittee is required to collect samples that are representative of their discharges and follow specified analytical methods. Failure to comply with any of these requirements is a violation of the Clean Water Act.

Self-monitoring Requirements

The permit must prescribe the self-monitoring procedures, frequency of analysis, and the locations and procedures for sample collection. Permit conditions require that analytical methods referenced in 40 CFR 136 be used for analysis. When methods are not included in Part 136, the permit must specify the methodology to be followed. Most self-monitoring data are reported on a Discharge Monitoring Report (DMR). Permittees are required to maintain all records associated with monitoring for a period of three years.

The frequency of monitoring is the most important factor in determining compliance with the permit limitations. As explained in the previous section, permit limits are set quite high relative to the required LTA performance. Therefore, sampling and analysis must be sufficiently frequent to detect violations of permit limitations at a non-complying facility. On the other hand, monitoring can be costly — particularly for toxic materials. The amount of monitoring that should be required is a function of how much variability is expected for the pollutant parameter, the compliance history of the facility, and the potential for water quality impacts due to non-compliance.

As a general guideline, if the value of a parameter is limited by the permit, the monitoring frequency should be at least once per month. For most toxic pollutants and effluent toxicity, monitoring once per week or more frequently may be needed because water quality impacts may result from non-compliance. Examples of monitoring frequencies are discussed in Section 8.

The total cost of toxic pollutant monitoring could be reduced by requiring a high initial monitoring frequency that can be decreased if the permittee consistently meets the limit. The overall purpose of such requirements is to establish a compliance history with a higher monitoring frequency during an initial period and to reduce the monitoring frequency for facilities that routinely comply with permit conditions. The requirements should be pre-specified in the permit and should take into consideration the basis of the permit limitations. For example, if weekly monitoring shows the permittee complies with the limitation, monitoring frequency may be reduced to once per month. Another approach is to concentrate monitoring frequency during critical periods to protect water quality.

Quality assurance (QA) is an important element of self-monitoring. If the referenced testing methodology includes QA procedures, the permit should require compliance with the procedures. If the methodology contains inadequate QA procedures, the permit should reference and require other QA procedures or set forth QA procedures in the permit. When QA standards are not met, the analysis should be repeated. EPA and many States also maintain laboratory QA programs. These programs can include laboratory inspections, "blind" sample analysis, and laboratory certification.

Inspections

Inspections are conducted to verify permittee compliance status determined from self-monitoring data. Compliance inspections may include reviewing records, inspecting treatment facilities, evaluating laboratory facilities and performance, and collecting samples for analysis. EPA has defined several types of inspections based on the tasks that are included [1]. The inspections that focus on toxics control, Compliance Sampling Inspections for toxics, and Compliance Biomonitoring Inspections, can also provide useful information for water quality assessment. Therefore, the regulatory authority should consider coordinating these inspections with water quality assessments and permit reissuance activities. Procedures for inspecting toxicity testing facilities and measuring effluent toxicity are detailed in the Compliance Biomonitoring Inspection Manual [2].

Regulatory authorities should also consider requiring the use of ambient toxicity testing, described in Appendix C, or other ambient testing as a monitoring tool to determine if the receiving water is being protected, i.e., the designated use is maintained or restored.

Enforcement

Regardless of the statistical basis used to develop a permit limit, exceeding the limit is a violation. The permittee must advise the regulatory agency of any instance in which a permit limit is exceeded in a written non-compliance report within five days after the violation is detected. This report must include a description of the non-compliance and its cause, the period of non-compliance, and if the non-compliance has not been corrected, the anticipated time it is expected to continue, and steps taken to reduce, eliminate, and prevent recurrence of the non-compliance. New regulations for reporting instances of non-compliance, including those related to water quality-based controls, have been promulgated (50 FR 34648, Monday, August 26, 1985).

If the permittee suspects analytical problems may have contributed to the violation, the permittee may perform additional analyses. The results of any additional analyses must also be reported. The regulatory agency should review the non-compliance report, request additional information from the permittee if it is needed, and review other relevant compliance data when an instance of non-compliance is detected. The regulatory agency decision to initiate enforcement actions, conduct inspections, establish a compliance schedule, or continue review of the permittee's self-monitoring should be made consistent with the principles of EPA's Enforcement Response Guide [3].

In some cases where the regulatory agency believes that a permittee's discharge may be causing an adverse water quality impact, a 308 letter can be used to obtain additional information and monitoring from the permittee. If such information shows that a permittee has failed to meet permit conditions or effluent limits but the type of remedy needed is unclear, the regulatory agency may establish a compliance schedule for further testing, evaluation of a remedy, and implementation of additional treatment. This compliance schedule could be implemented by an administrative order or consent decree. For example, a compliance schedule may require that the permittee conduct biological and chemical analysis several times during the year before a permit is reissued. The compliance schedule should identify triggers or thresholds that would require further monitoring or corrective actions, define or reference the types of analysis required, and establish criteria that would indicate whether further controls and limits were needed. The schedule should also include milestones for reporting and providing additional toxic controls if they are needed.

References

1. U.S. Environmental Protection Agency. 1984. NPDES Compliance Inspection Manual. OWEP. June NTIS # PB85-115897.
2. U.S. Environmental Protection Agency. 1979. Interim NPDES Compliance Biomonitoring Inspection Manual. MCD-62. Office of Water Enforcement (EN-338), Washington, D.C.
3. U.S. Environmental Protection Agency. 1985. Enforcement Management System. Office of Water Enforcement and Permits, Washington, D.C.

8. *Case Example*

In this section, an example of a water quality-based discharge situation is described to illustrate the use of the procedures detailed in the previous sections. All of the recommendations and options are not incorporated in the example. Although the site is fictional, much of the data have been taken from previously conducted studies.

Site Description

The Linville River is a small to medium sized river flowing through an area of mixed land use. The river is 60 miles long and its banks are typified by small towns separated by wooded areas. The 7Q10 of the river above the first discharger is 10.7 million gallons per day (MGD) (16.6 cfs). From its monitoring programs, the State had indications that the water quality of the middle section was somewhat degraded. The dischargers located in this section (starting from the farthest upstream) are Cibesi Corporation, a metal finishing plant and foundry, Nicklaus By-Products Company, a coal tar extractor producing a variety of organic solvents, and the town of Bibury sewage treatment plant (STP). All other sources in the area discharge into the STP. NPDES permits for both industrial sources (which the State permit writers had previously issued with Best Professional Judgment [BPJ] Best Available Technology [BAT] permit limits) had expired, and the STP permit was coming up for renewal.

The location and a description of each discharger is provided below:

River Mile 22.1 Cibesi Corporation
Manufacturing (SIC 3362 + 3325 Foundry; 3471 Electroplating)
Treatment: equalization; neutralization; settling.
Discharge Flow: 0.5 MGD process wastes
Dilution at 7Q10 (assuming complete mixing): 4.5%

River Mile 21.4 Nicklaus By-Products
Coal Tar Extractor (SIC 2865)
Treatment: aerated lagoon and chlorination
Discharge Flow: 3.0 MGD process and sanitary wastes
Dilution at 7Q10 (assuming complete mixing): 21%

River Mile 13.8 Bibury STP
Treatment: Activated sludge, Phosphorus removal
Discharge Flow: 2.3 MGD average (1 MGD industrial input)
Dilution at 7Q10 (assuming complete mixing): 14%

Criteria Determination (Section 2)

Linville River is classified for the propagation of aquatic life and is used for a variety of recreational activities. The State reviewed available data to identify pollutants that may be causing water quality impacts. The following pollutants were identified: cyanide (CN), phenols, copper (Cu), cadmium (Cd), nickel (Ni), chromium (Cr^{+6}), and zinc (Zn). Assorted organic priority pollutants, including an array of chlorinated phenolic compounds, had been measured at Nicklaus By-Products on several occasions. However, presence and concentrations were extremely variable.

The State had conducted a trial evaluation of site-specific criteria modification and had adopted the EPA criteria for copper and cyanide as State standards. No other toxicants were added to the State's numerical water quality standards so EPA criteria were also used for the other toxicants in this study.

Pollutant	*Criteria Concentrations* (at 100 mg/l hardness)* CMC (μg/l)	CCC (μg/l)
CN	22.0	4.2
Cu	16.0	11.0
Cr^{+6}	11.0	7.2
Cd	4.5	4.5
Ni	1,800.0	96.0
Zn	320.0	47.0
Phenols	—	—

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* These numbers are based on previous EPA criteria values.

EPA criteria are available only for specific phenolic compounds, so the State did not have a recommended value for the total phenols measured in this study.

The State applied the numerical criteria described in Section 2. Thus, each discharger must meet the criteria of 0.3 acute toxic units (TU_a) within the mixing zone and 1.0 chronic toxic units (TU_c) at the edge of the mixing zone.

The State also examined the question of duration and frequency criteria for the site. State water quality standards are one-number chronic values. State regulations specify that the standards apply only at flows equal to or greater than the 7Q10. EPA recommended that two wasteload allocations (WLAs) be performed; one for the acute criterion and one for the chronic criterion. Since one-day duration is more appropriate for acute criteria than seven days, the State decided to use a 1Q10 for the acute WLA and the state-specified 7Q10 for the chronic WLA.

Effluent Characterization (Section 3)

Chemical Data Available (in micrograms/liter [μg/l])

	Cibesi Corporation		Nicklaus By-Products	
Pollutant	DMR*	Form 2C	DMR*	Form 2C
CN total	530.0	2300.0	210.0	420.0
Cu	1130.0	2130.0	—	—
Cr^{+6}	290.0	215.0	—	—
Cd	210.0	660.0	—	—
Ni	1130.0	3130.0	—	—
Zn	620.0	2560.0	—	—
Phenol (4AAP)	—	—	1200.0	6120.0

* Two-year average of reported monthly averages from the DMRs.

Chemical screening using simple dilution calculations showed that the pollutants present in two of the three effluents could be exceeding water quality criteria values at various times after mixing. Chemical data for the Bibury STP showed only low levels of metals. Due to the complex nature of the effluents involved and the lack of adequate effluent toxicity data, the regulatory authority concluded that effluent toxicity analysis was necessary. For Cibesi, a mixture of both cyanide and five metals at potentially toxic levels suggested a whole-effluent toxicity assessment. An analysis of instream concentrations of copper and cyanide also seemed appropriate since the State had numerical standards for these toxicants and since they were present at very high levels relative to their effect levels. For Nicklaus By-Products, the presence of high levels of phenols (for which no chemical-specific characterization is feasible) suggests whole-effluent toxicity analysis is the most efficient way to assess toxic impact. For the STP, low dilution capacity also suggested toxicity testing even though available chemical data had not indicated any specific problems.

Initial Permit Limits

The State calculated initial toxicity-based permit limits for Cibesi and Nicklaus By-Products using the equation:

$$\text{allowable effluent toxicity} \le \text{criterion} \times \text{dilution factor}$$

For Cibesi, the initial limits were 4.8 TU_a and 19.3 TU_c. For Nicklaus, the initial limits were 0.4 TU_a and 1.5 TU_c. Derivation of these limits is explained in the following subsection entitled Effluent Toxicity Wasteload Allocation. Both companies objected to the establishment of toxicity-based permit limits, arguing that they were meeting their BAT permit limits, they were being given water quality limits for some specific chemicals, and no toxic impact caused by their discharges had been proved. The State responded that the nature of their discharges, particularly those of Nicklaus, made toxicity assessment mandatory due to the low dilution available, the number of toxicants known to be present in their discharges, and the legal requirement to enforce the "no toxics in toxic amounts" criterion in the State standards.

After reviewing the situation, each company agreed to conduct toxicity testing. Specific testing procedures were determined as follows.

Tier 1 — Toxicity Screening (Section 3)

The regulatory authority determined on the basis of dilution alone that a strong potential exists for water quality standards violation (toxic impact) for each discharge. Accordingly, the State's normal toxicity screening procedures were bypassed, and definitive toxicity data procedures were implemented for each permittee.

Tier 2 — Definitive Data Generation — Toxicity (Section 3)

Requirement
The three permittees were given the same definitive data generation requirements via §308 letters. They were required to conduct effluent toxicity tests following a tiered toxicity testing procedure as described in the U.S. EPA Technical Support Document (Office of Water, September, 1985). Dilution available at low-flow periods on the Linville River falls within the effluent-dominated category of receiving waters. Therefore, chronic toxicity data will be required.

Uncertainty factors to be applied:

- For species sensitivity
  - 10X, where two test species are used.
  - 1X, where three test species are used.
- For effluent variability:
  - 100X, where testing is limited to one test per test species quarterly.
  - 10X, where testing is limited to two tests per test species monthly.
  - 1X, where testing conducted on four samples per test species monthly.

The regulatory authority will determine further data generation requirements when the uncertainty factors analysis is made on the basis of the following comparison at each test tier:

$$\frac{\text{NOEL}}{\text{IWC}} > \text{level of uncertainty}$$

where IWC is the instream waste concentration and NOEL is the No Observed Effect Level. If, after these tests, the regulatory authority determines no more data are warranted due to a large difference between effect level and level of uncertainty, testing can be terminated. Where no such difference exists, further testing requirements will be determined.

Special Considerations

Several special considerations have been provided by the State to reduce testing costs, where appropriate:

- If three species are tested, only the most sensitive species need be tested after the first two months of testing identifies that species.
- If the difference between each monthly sample's NOEL is less than 50% after the first two months of testing, the sampling frequency can be reduced by half.
- Where greater than 100% effect levels are observed for all tests after the first two months of testing, data generation procedures can be terminated.

Definitive Data Generation Requirements for Each Example Permittee

Cibesi Corporation

TESTING

Based on dilution calculations, the company has an IWC of 4.5%. If they conducted chronic toxicity tests and consistently measured NOELs of greater than 100% (no observable toxicity) their best NOEL/IWC ratio would be:

$$\frac{> 100\%}{\frac{0.5\text{ MGD}}{10.7\text{ MGD} + 0.5\text{ MGD}} \times 100} = 22$$

They would be able to pass successfully a testing tier using an uncertainty factor of 10 (22 is greater than 10) but not a testing tier with an uncertainty factor of 100 or greater. Bearing in mind the testing costs and the special considerations described above, Cibesi negotiated a testing requirement (Tier 2 testing) of four tests (on consecutive-day samples) per species conducted over a two-month period using *Ceriodaphnia* and the fathead minnow. The company and the State agreed that this testing procedure was the most efficient since the results of the first two months of testing would dictate whether or not additional data generation would be required.

RESULTS

	Sample			
January, 1983	1	2	3	4
Ceriodaphnia NOEL	10%	25%	1%	5.5%
Fathead minnow NOEL	70%	>100%	50%	>100%

These first months' results surprised the company and pointed to the probable need for toxicity reduction procedures. At the lowest toxicity level (1%) no uncertainty factor comparison could be passed (NOEL/IWC is less than one). The regulatory authority determined that a complete toxicity assessment was required to establish permit limitations. A testing procedure was established to provide a more complete effluent characterization: conduct four tests (samples collected on consecutive days) on three species (must include *Ceriodaphnia*) monthly (or every two months) for a total of at least six months. The special considerations listed above will apply. Accordingly, the following data were generated:

	Sample			
February, 1983	1	2	3	4
Ceriodaphnia NOEL	10%	25%	5.5%	1%
Fathead minnow NOEL	70%	>100%	50%	>100%
Selenastrum NOEL	>100%	>100%	>100%	>100%
April, 1983	1	2	3	4
Ceriodaphnia NOEL	5.5%	1%	5.5%	1%
Fathead minnow NOEL	50%	25%	NA*	70%
Selenastrum NOEL	>100%	>100%	>100%	>100%
June, 1983	1	2	3	4
Ceriodaphnia NOEL	5.5%	NA*	1%	1%
Fathead minnow NOEL	dropped as a requirement; not sensitive			
Selenastrum NOEL	dropped as a requirement; not sensitive			
	1	2	3	4
Ceriodaphnia only:				
August, 1983	1%	1%	5.5%	10%
September, 1983	5.5%	NA	10%	8%

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* Not available (test failed).

These data were sent to the regulatory authority by the Cibesi Corporation where they were used to establish a WLA.

Nicklaus By-Products

TESTING

The company had an estimated IWC of 21.0%. If they conducted chronic toxicity tests and consistently measured NOELs of greater than 100% (no observable toxicity) their best NOEL/IWC ratio would be:

$$\frac{> 100\%}{\dfrac{3.0 \text{ MGD}}{10.7 \text{ MGD} + 0.5 \text{ MGD} + 3.0 \text{ MGD}} \times 100} = 4.7$$

They would not be able to pass sucessfully any tiered testing that applied an uncertainty factor (since 4.7 is less than 10, the minimum uncertainty factor). This would require the company to implement a testing program that would generate a complete database. No tiered assessment utilizing some uncertainty factor or combination of factors would be allowed for this discharge situation. Therefore, the company proposed the following testing requirement: conduct chronic toxicity tests on three species (*Ceriodaphnia, Pimephales,* and *Selenastrum*) using four samples taken on consecutive days per month for a total of at least six months. The special considerations described above should apply. The company requested that it be allowed to conduct an analysis of persistence and instream degradation if toxicity were measured. The regulatory agency agreed and revised the requirement to say that if the first month's testing showed toxicity, the company would be allowed to analyze the persistence of the toxicity measured. Accordingly, the following data was generated:

	Sample			
January, 1983	1	2	3	4
Ceriodaphnia NOEL	50%	>100%	10%	70%
Fathead minnow NOEL	5.5%	10%	<1%	1%
Selenastrum NOEL	10%	50%	20%	50%

The extreme toxicity of the effluent, particularly to fathead minnow, pointed to the need for toxicity reduction procedures. However, the company argued that phenolic compounds were probably causing the toxicity. These compounds, according to Nicklaus By-Products, were not persistent after discharge and lost their toxic effect after mixing. The company cited the lack of fish-kills and obvious stream degradation as evidence that toxicity was non-persistent.

The regulatory authority cited two procedures for persistence analysis the company could use: shelf aging and ambient toxicity testing. Further, the company was given the option of conducting a biosurvey to assess instream degradation of the biota in the river. The company first conducted tests following a sample aging procedure in which an effluent sample was held for 24, 48, and 96 hours on the shelf prior to toxicity testing with the following results:

Pimephales promelas seven-day growth test:

24-hour-old sample NOEL = 1.0%
48-hour-old sample NOEL = 5.5%
96-hour old sample NOEL = 8.0%

These results were inconclusive. Next, the company conducted ambient toxicity testing following procedures detailed in EPA's Technical Support Document (Appendix C). A dye study of effluent dispersion is required at a design low-flow period to determine mixing characteristics.

The dye study showed the effluent rapidly mixed after discharge. The effluent reached equilibrium (IWC = 30%) approximately 500 feet below the discharge point.

Next, six ambient stations were established along the river:

Station		Location
1	RM 31.2	Upstream (headwaters) control
	RM 22.1	(Cibesi Corporation discharge)
2	RM 21.4	Directly upstream of discharge
	RM 21.4	(Nicklaus By-Products discharge)
3	RM 21.4	Directly downstream of discharge
4	RM 21.5	At point of complete mixing
5	RM 20.4	One mile downstream
6	RM 16.4	Five miles downstream

Two test species were used with the following results:

Species	Stream Station					
	1	2	3	4	5	6
Ceriodaphnia						
(# young/female)	24.1	10.4	5.5	8.0	9.1	16.0
(% survival)	100	60	20	40	60	90
Fathead minnow						
(mean weight in mg)	0.47	0.40	–	–	–	0.31
(% survival)	90	90	0	0	0	30

Ambient toxicity results showed a pattern of instream toxicity that would be expected if effluent toxicity data were compared to effluent concentrations instream. At Station 1, both species showed standard growth and reproduction. At Station 2, below Cibesi, no effect was seen for fathead minnows, but the *Ceriodaphnia* were somewhat affected (as expected given instream concentrations of their effluent and the sensitivity of the organism to the Cibesi discharge). Samples from station 3, 4, and 5 below Nicklaus, showed lethality to the minnows. Mortality was observed at the point of discharge, at complete mixing, and at one mile downstream, respectively. Complete recovery for fathead minnows was not observed even at Station 6, five miles downstream. *Ceriodaphnia* did show recovery at this station. Toxicity for Nicklaus By-Products was thus considered persistent by the regulatory authority at the low-flow period observed.

The State determined that Nicklaus By-Products was impacting the Linville River. The State then required the company to generate additional effluent toxicity data (following the procedures previously described) so that appropriate limits could be calculated.

RESULTS

February, 1983	Sample 1	2	3	4
Ceriodaphnia NOEL	---- excessive control mortality ----			
Fathead minnow NOEL	1%	10%	5.5%	1%
Selenastrum NOEL	50%	>100%	>100%	>100%

March, 1983	1	2	3	4
Ceriodaphnia NOEL	100%	>100%	>100%	>100%
Fathead minnow NOEL	50%	70%	30%	10%
Selenastrum NOEL	>100%	>100%	>100%	>100%

April, 1983	1	2	3	4
Fathead minnow NOEL	10%	5.5%	2.5%	5.5%
Ceriodaphnia NOEL	dropped as a requirement; not sensitive			
Selenastrum NOEL	dropped as a requirement; not sensitive			

	1	2	3	4
Fathead minnow only:				
May, 1983	5.5%	10%	1%	1%
June, 1983	10%	5.5%	3%	1%

These data were presented to the regulatory authority. The State will use the data to develop a WLA.

Bibury STP

TESTING

Based on dilution calculations, Bibury STP has an IWC of 14%. If the STP conducted chronic toxicity tests and consistently measured NOELs of 100%, their best NOEL/IWC ratio would be:

$$\frac{> 100\%}{\dfrac{2.3 \text{ MGD}}{(10.7 + 0.5 + 3.0 + 2.3) \text{ MGD}} \times 100} = 7.2$$

They would not be able to pass successfully a testing tier using any uncertainty factor (since 7.2 is less than 10, the minimum uncertainty factor). This would require the STP to implement a testing program that would generate a complete database. No tiered assessment utilizing some uncertainty factor or combination of factors will be allowed for this discharge situation. Therefore, the STP agreed to conduct the following testing: run chronic toxicity tests on three species (*Ceriodaphnia*, *Pimephales*, and *Selenastrum*) using four samples taken on consecutive days over a six-month period. The special considerations previously described will apply. The STP began testing with the following results:

February, 1983	Sample 1	2	3	4
Ceriodaphnia NOEL	>100%	>100%	>100%	>100%
Fathead minnow NOEL	>100%	>100%	>100%	>100%
Selenastrum NOEL	>100%	>100%	>100%	>100%

April, 1983	1	2	3	4
Ceriodaphnia NOEL	>100%	>100%	>100%	>100%
Fathead minnow NOEL	>100%	>100%	>100%	>100%
Selenastrum NOEL	>100%	>100%	>100%	>100%

Since no chronic toxicity was observed during the first two months of testing, the regulatory authority determined that the Bibury STP effluent was not toxic, and further testing was terminated. No further water quality-based limitation derivation procedures were conducted.

Tier 2. — Definitive Data Generation — Chemical-specific (Section 3)

Cibesi Corporation and Nicklaus By-Products were required to conduct two plug-flow sampling programs for cyanide during low flows approximately equal to the WLA design flows of 1Q10 and 7Q10. In addition, Cibesi was required to include copper in their plug-flow sampling program. Cyanide was selected for Tier 2 monitoring because the State has numerical water quality standards for this pollutant and it is discharged at potentially acutely toxic levels by both Cibesi and Nicklaus. Monitoring was required for copper because the State has numerical water quality standards for this pollutant. DMR and 2C data indicated that this metal is discharged from Cibesi at concentrations significantly above acutely toxic levels. Monitoring and modeling of phenols is not required since many forms of this compound may be present and each one would require identification for comparison to EPA criteria. Whole-effluent toxicity analyses should result in the control of phenols as well as the other toxic compounds without requiring detailed pollutant-specific analyses of each chemical.

Data from one sampling period are needed for WLA model calibration. A second sampling period is then required for model verification. Flow conditions should be similar for both sampling events so that calibrated settling and resuspension rates for copper will be transferable to the verification period. Pollutant loadings from Cibesi and Nicklaus must vary during the two periods so that verification is a good test of the calibrated rate coefficients.

Requirement

The firms were required to conduct two plug-flow sampling analyses by following a dye slug along the Linville River. Plug-flow sampling involves the collection of water quality samples at each station and each effluent as the leading edge and peak of the dye slug is passing that point.

Results

Seven instream stations were established along the river:

Station		Location
1	RM 22.2	Directly upstream of Cibesi
	RM 22.1	(Cibesi Corporation discharge)
2	RM 22.0	Downstream of Cibesi at point of complete mixing
3	RM 21.65	0.35 miles downstream of Cibesi
4	RM 21.45	0.7 miles downstream of Cibesi (directly upstream of Nicklaus)
	RM 21.4	(Nicklaus By-Products discharge)
5	RM 21.3	Downstream of Nicklaus at point of complete mixing
6	RM 17.6	3.8 miles downstream of Nicklaus
7	RM 13.8	7.6 miles downstream of Nicklaus (directly upstream of Bibury STP)

Tables 8-1 and 8-2 show the results of the two sampling programs.

Table 8-1. Plug Flow Sampling #1

	Stream Station Data						
Constituent	1	2	3	4	5	6	7
Total CN (ug/1)	0	26.8	25.7	24.6	66.4	41.9	26.3
Dissolved CN (ug/1)	0	26.8	25.7	24.6	66.4	41.9	26.3
Total Cu (ug/1)	0	57.2	56.1	55.1	42.7	34.6	29.2
Particulate Cu (ug/1)	0	31.9	30.8	29.8	25.3	17.2	11.8
Dissolved Cu (ug/1)	0	25.3	25.3	25.3	17.4	17.4	17.4
TSS (mg/1)	20	21	20.3	19.6	24.2	16.5	11.3
pH	7	7	7	7	7	7	7
Temperature (°C)	20	20	20	20	20	20	20
Hardness (mg/1 $CaCO_3$)	100	100	100	100	100	100	100
Flow (cfs)	15	15.8	15.8	15.8	20.4	20.4	20.4
Velocity (ft/sec)	0.5	0.5	0.5	0.5	0.5	0.5	0.5

Effluent Data

	Cibesi	Nicklaus
Total CN (ug/1)	530	210
Total Cu (ug/1)	1130	–
TSS (mg/1)	40	40
Flow (cfs)	0.8	4.6

Table 8-2. Plug Flow Sampling #2

	Stream Station Data						
Constituent	1	2	3	4	5	6	7
Total CN (ug/1)	0	124.3	119.1	114.1	186.6	117.8	74.4
Dissolved CN (ug/1)	0	124.3	119.1	114.1	186.6	117.8	74.4
Total Cu (ug/1)	0	115.1	112.9	110.9	84.6	68.8	58.0
Particulate Cu (ug/1)	0	58.0	56.8	53.8	49.8	34.0	23.2
Dissolved Cu (ug/1)	0	57.1	57.1	57.1	34.8	34.8	34.8
TSS (mg/1)	15	16.9	16.3	15.7	23.8	16.3	11.1
pH	7	7	7	7	7	7	7
Temperature (°C)	20	20	20	20	20	20	20
Hardness (mg/1 $CaCO_3$)	100	100	100	100	100	100	100
Flow (cfs)	14	14.8	14.8	14.8	19.4	19.4	19.4
Velocity (ft/sec)	0.5	0.5	0.5	0.5	0.5	0.5	0.5

Effluent Data

	Cibesi	Nicklaus
Total CN (ug/1)	2300	420
Total Cu (ug/1)	2130	–
TSS (mg/1)	50	50
Flow (cfs)	0.8	4.6

Exposure Assessment (Section 5)

Effluent Toxicity Wasteload Allocation

The dye study showed that mixing is rapid and complete. Therefore, a mixing zone analysis is not required to assess exposure, and the State used a complete mix WLA approach. A steady state WLA for effluent toxicity was performed for Cibesi Corp. and Nicklaus By-Products. As explained in Section 3, the analysis is based on the assumption that toxicity is conservative and additive. The ambient toxicity data collected by Nicklaus By-Products indicate that this assumption is reasonable.

The input data required for the WLA include: the chronic criteria, effluent flow, ambient toxicity upstream from Cibesi, and the critical design flow of the Linville River. The instream chronic criterion is one toxic unit, measured as chronic toxicity to *Ceriodaphnia* (the most sensitive species to the Cibesi effluent) or fathead minnows (the most sensitive species to the Nicklaus By-Products effluent). Criteria for duration and recurrence are as described above. Effluent flow is given as 0.5 MGD for Cibesi and 3.0 MGD for Nicklaus By-Products. Toxicity upstream from Cibesi is assumed to be zero since there are no significant dischargers above this point and ambient toxicity data collected by Nicklaus By-Products showed good water quality (no measureable toxicity) at the control station above the discharge points.

The toxicity effects of both effluents are assumed to be comparable so that wasteloads can be allocated between them even though two different test organisms turned out to be most sensitive. The rationale is that both species merely represent the more sensitive end of the range of potential sensitivities. The measured toxicity from each effluent will ''overlap'' for the many species in the receiving water. The toxic effects measured by *Ceriodaphnia* and minnows can be assumed comparable for the purpose of developing a toxicity WLA (see Section 1).

Since the State water quality standards specify that they apply only at or above the 7Q10, this flow is selected as the chronic design flow for Linville River. A 1Q10 is selected as the acute design flow since the one-day period is more appropriate for acute event duration (see Appendix D).

A U.S. Geological Survey (USGS) flow gauging station is located on the Linville River directly above the Cibesi outfall. The low-flow statistics program in STORET calculates the 1Q10 at this station as 8.9 MGD or 13.7 cfs and the 7Q10 as 10.7 MGD or 16.6 cfs.

This multiple-discharge WLA is based on flow-weighting of the assimilative capacity. There are many other ways of doing a multiple-source WLA (e.g., allocating the total allowable load based on flow proportion, cost parity, etc.). Cibesi will be allowed to use 86% of the assimilative capacity of the river since Cibesi flow is only 14% of the total point source flow to the river. Consequently, the instream toxicity below Cibesi must equal 0.86 times the criterion for both acute and chronic effects. Nicklaus By-Products will be required to provide the remainder of the treatment needed to maintain the criteria for toxic effects.

The WLAs for *acute* protection were developed using an acute-chronic ratio (ACR) of 10, which was determined from the chronic toxicity testing data by looking at 48 versus the 168 hour (seven-day) NOEL. The effluent toxicity required for Cibesi to meet the maximum acute criterion is calculated as follows:

$$\frac{(C_sQ_s) + (C_eQ_e)}{Q_e + Q_s} \leq (0.86) \text{ Maximum Acute Criterion}$$

where C_s = upstream TU_a
Q_s = acute critical flow (1Q10)
C_e = required effluent TU_a
Q_e = effluent flow
Maximum Acute Criterion = 0.3 TU_a

$$\frac{(0)\,(8.9) + (C_e)\,(0.5)}{0.5 + 8.9} \leq (0.86)\,(0.3)$$

$$C_e \leq 4.85\ TU_a$$

or using the ACR,

$$C_e \leq 48.5\ TU_c$$

The effluent toxicity required for Nicklaus By-Products to meet the maximum acute criterion is calculated as follows:

$$\frac{(0.258)\,(9.4) + (C_e)\,(3.0)}{9.4 + 3.0} \leq (0.3)$$

$$C_e \leq 0.43\ TU_a$$
$$C_e \leq 4.3\ TU_c$$

The effluent toxicity required for Cibesi to meet the chronic requirement is calculated as follows:

$$\frac{(C_sQ_s) + (C_eQ_e)}{Q_e + Q_s} \leq (0.86) \text{ Chronic Criterion}$$

where C_s = upstream TU_c
Q_s = chronic critical flow (7Q10)
C_e = required effluent TU_c
Q_e = effluent flow

Chronic Criterion = 1.0 TU_c

$$\frac{(0)\,(10.7) + (C_e)\,(0.5)}{0.5 + 10.7} \leq 1(0.86)$$

$$C_e \leq 19.3\ TU_c$$

The effluent toxicity required for Nicklaus By-Products to meet the chronic requirement is calculated as follows:

$$\frac{(0.86)\,(11.2) + (C_e)\,(3.0)}{11.2 + 3.0} \leq 1$$

$$C_e \leq 1.5\ TU_c$$

Pollutant-specific WLA

At Stations 2 and 5, the WLA model used the complete mix mass balance equations presented above. At the other stations the equation for cyanide consisted of the following:

$$CN_x = CN_u\, e^{-kx/u}$$

where CN_x = cyanide concentration at distance x (µg/l)
CN_u = upstream cyanide concentration (µg/l)
k = first order decay rate (l/day)
x = downstream distance (ft)
u = stream velocity (ft/sec)

The equations for copper at the other stations consisted of the following:

$$TSS_x = TSS_u\, e^{-W_s/H(x/u)}$$

$$Cu_t = Cu_d\,(1 + K_pTSS_x)$$

$$Cu_p = Cu_t - Cu_d$$

where TSS_x = total suspended solids concentration at distance x (mg/l)
TSS_u = upstream total suspended solids concentration (mg/l)
W_s = settling velocity of TSS (ft/day)
H = depth of water (ft)
x = downstream distance (ft)
u = stream velocity (ft/sec)
Cu_t = total copper concentration at distance x (µg/l)
Cu_d = dissolved copper concentration at distance x (µg/l)
Cu_p = particulate copper concentration at distance x (µg/l)
K_p = copper partition coefficient (l/kg)

The following rate coefficients were calibrated and verified with the data from the two sampling events:

K = l/day for cyanide
W_s = 0.82 ft/day
K_p = 60,000 l/kg

The calibrated constants were within the range reported in the technical literature. Resuspension was not observed during low flow so modeling of TSS and Cu did not include this term. The WLA model described above was used with the calibrated rate coefficients and the 1Q10 flow to calculate the allowable acute loads for copper and cyanide. The model was also used with the 7Q10 flow to calculate the allowable chronic loads. The same flow-weighting rule used for the multiple-discharge WLA of generic toxicity was also used for cyanide. Tables 8-3 and 8-4 show the predicted instream results of acute and chronic WLAs for copper and cyanide.

The following assumptions were made for the WLA:

- Upstream of Cibesi TSS = 15 mg/l.
- Upstream of Cibesi CN and Cu equal zero.
- Stream velocity at all stations is 0.5 ft/sec (as indicated by monitoring data).
- Depth of flow at all stations is one foot (as indicated by the monitoring data).
- New treatment will result in Cibesi and Nicklaus effluent TSS = 30 mg/l.

The 1985 EPA national criteria for copper specify the "active" form of the metal. Acid-soluble copper is distinguished from other forms by the analytical technique used to measure it. Acid-soluble copper is always less than total copper and greater than dissolved copper. The WLA done for this situation used the conservative assumption that total copper must be equal to the acute and chronic criteria at the appropriate critical flows. The 1985 cyanide criteria specify "free cyanide." Since all the cyanide in this situation is dissolved, all the chemical is considered to be in the free form.

In order to meet the 1985 instream criteria for copper and cyanide with flows of 0.5 MGD at Cibesi and 3.0 MGD at Nicklaus, the following loads are allowable:

Cyanide WLA:	Cibesi	Nicklaus
Acute one-day CN load (lb/day)	1.59	0.74
Acute one-day CN concentration (μg/l)	370	30.0
Chronic seven-day CN load (lb/day)	0.37	0.15
Chronic seven-day CN concentration (μg/l)	85.0	6.0
Copper WLA:		
Acute one-day Cu load (lb/day)	1.25	—
Acute one-day Cu concentration (μg/l)	290	—
Chronic seven-day Cu load (lb/day)	1.03	—
Chronic seven-day Cu concentration (μg/l)	239	—

Table 8-3. Acute WLA Results

	Stream Station						
Constituent	1	2	3	4	5	6	7
Total CN (ug/1)	0	20.4	19.5	18.7	21.4	13.5	8.5
Dissolved CN (ug/1)	0	20.4	19.5	18.7	21.4	13.5	8.5
Total Cu (ug/1)	0	16.0	15.7	15.5	11.8	9.9	8.5
Particulate Cu (ug/1)	0	7.8	7.5	7.3	6.2	4.3	2.9
Dissolved Cu (ug/1)	0	8.2	8.2	8.2	5.6	5.6	5.6
TSS (mg/1)	15	15.8	15.3	14.8	18.5	12.7	8.7

Table 8-4. Chronic WLA Results

	Stream Station						
Constituent	1	2	3	4	5	6	7
Total CN (ug/1)	0	3.9	3.7	3.5	4.0	2.5	1.6
Dissolved CN (ug/1)	0	3.9	3.7	3.5	4.0	2.5	1.6
Total Cu (ug/1)	0	11.0	10.9	10.7	8.5	7.1	6.1
Particulate Cu (ug/1)	0	5.3	5.2	5.0	4.4	3.0	2.0
Dissolved Cu (ug/1)	0	5.7	5.7	5.7	4.1	4.1	4.1
TSS (mg/1)	15	15.7	15.2	14.7	17.9	12.2	8.3

Permit Limit Derivation (Section 6)

The method for deriving permit limitations is a result of the way that the WLA for the discharger is expressed. In this example, the WLAs for both dischargers were developed using the acute and chronic steady state approach. This approach yields requirements expressed as a one-day value and a seven-day average value for effluent quality. To derive limits, the State permit writer had to determine which requirement was more limiting and to derive daily maximum and average daily limits to enforce that requirement properly. To illustrate the calculations, the derivation of limits for the copper WLA for Cibesi is described in detail below. The outcome for the other WLAs is summarized in Tables 8-5 and 8-6 and described in general.

The WLA requirements for copper were specified as follows: the one-day concentration should be less than 290 μg/l, and the seven-day average concentration should be less than 239 μg/l. The State determined that the occurrence probability for the WLA values should be 0.01. They estimated that the CV for copper after compliance would be 0.60 by looking at the development documents for metal finishing and electroplating.

The log variance, σ, was calculated as follows:

$$\begin{aligned} \sigma^2 &= \ln(CV^2 + 1) \\ &= \ln(0.60^2 + 1) \\ &= 0.3075 \\ \sigma &= \sqrt{0.3075} = 0.5545 \end{aligned}$$

The z value corresponding to a probability of 0.01 is 2.326.

The required long term average (LTA) needed to meet the one-day requirement was calculated as follows:

$$\begin{aligned}\text{WLA} &= 290\ \mu\text{g/l}\\ \mu &= \ln(\text{WLA}) - Z\,\sigma\\ &= \ln 290 - 2.326\,(0.5545)\\ &= 4.3801\\ \text{LTA} &= e^{\mu + 0.5\sigma^2}\\ &= e^{4.3801 + 0.5(0.3075)}\\ &= 93\ \mu\text{g/l}\end{aligned}$$

The LTA needed to meet the seven-day requirement was calculated as follows:

$$\begin{aligned}\text{WLA} &= 239\ \mu\text{g/l}\\ \mu_7 &= \ln(\text{WLA}) - Z\sqrt{\ln[1+((e^{\sigma^2}-1)/7)]}\\ &= 4.9556\\ \mu &= \mu_7 - 0.5\,\sigma^2 + 0.5\ln[1+((e^{\sigma^2}-1)/7)]\\ &= 4.8269\\ \text{LTA} &= e^{\mu+0.5\sigma^2}\\ &= 145\ \mu\text{g/l}\end{aligned}$$

The LTA needed to meet the one-day WLA value is 93 μg/l. The LTA needed to meet the seven-day WLA value is 145 μg/l. The one-day or acute WLA requirement is more limiting for this plant. Therefore, permit limits were derived based on a required LTA of 93 μg/l.

To calculate permit limits, the permit writer needed to determine the probability basis for the permit limits (not to be confused with the probability basis for the WLA values) and the assumed frequency of observation for the monthly limits. For Cibesi, it was decided to require daily monitoring because of the potential water quality impact and because copper analyses are inexpensive. The compliance monitoring was considered sufficient to allow a permit limit probability basis of 0.01 instead of 0.05. The assumed number of observations for the monthly limit was n = 30. Permit limits were derived using Figure 6-3 as follows:

Daily maximum:

$$\begin{aligned}\mu &= \ln(\text{LTA}) - 0.5\,\sigma^2\\ &= \ln 93 - 0.5(0.3075)\\ &= 4.3801\\ \text{limit} &= e^{\mu+Z\sigma}\\ &= e^{4.3801+2.326\,(0.5545)}\\ &= 289\ \mu\text{g/l}\end{aligned}$$

Average daily (monthly):

$$\begin{aligned}\text{limit} &= \text{LTA}\,\left(1+\frac{Z}{\sqrt{30}}(\text{CV})\right)\\ &= 93\,\left(1+\frac{2.326}{\sqrt{30}}(0.60)\right)\\ &= 117\ \mu\text{g/l}\end{aligned}$$

A summary of the copper requirements can be seen in Table 8-5. In this case it was the one-day WLA requirement that was more limiting for the plant even though the WLA value was higher than the seven-day requirement. The daily maximum permit limit is the same as the WLA value only because the one-day WLA was limiting and because the probability basis for the WLA and the limits are the same.

Table 8-5. Cibesi Corporation

Wasteload Allocation			Required		Permit Limits	
Parameter	Duration	Value	L.T.A.	Basis	Monthly	Daily Maximum
Toxicity	1-day	48.5 tu_c	10.3 tu_c	95th	12.5 tu_c	21.8 tu_c
	7-day	19.3 tu_c	8.0 tu_c	n = 10		
CN	1-day	370 ug/l	69 ug/l	99th	52 ug/l	71 ug/l
	7-day	75 ug/l	36 ug/l	n = 30		
Cu	1-day	290 ug/l	93 ug/l	99th	117 ug/l	289 ug/l
	7-day	239 ug/l	146 ug/l	n = 30		

Table 8-6. Nicklaus By-Products

Wasteload Allocation			Required		Permit Limits	
Parameter	Duration	Value	L.T.A.	Basis	Monthly	Daily Maximum
Toxicity	1-day	4.3 tu_c	0.9 tu_c	95th	1.2 tu_c	1.9 tu_c
	7-day	1.5 tu_c	0.7 tu_c	—		
CN	1-day	30 ug/l	—	—	6 ug/l	12 ug/l
	7-day	6 ug/l				

Table 8-5 also describes Cibesi's requirements for toxicity and cyanide. The toxicity CV was estimated from the toxicity data collected during impact assessment. The value for CV was 0.96 and was assumed not to change. The CV for cyanide was obtained from the electroplating development documents and was estimated as 1.10. For both these parameters, the seven-day WLA requirement was limiting. The toxicity limits illustrate the greatest change needed to properly enforce the WLA. The factors that affect the limits are as follows:

- The WLA occurrence probability was set at 0.01, but the permit limit basis was 0.05 because monitoring would be infrequent (once a week).
- The seven-day WLA was the limiting requirement. If daily effluent values were allowed to go to the one-day WLA value, non-compliance with the seven-day WLA requirement is projected.
- The chronic WLA value is a seven-day average, whereas the monthly permit limit was derived as a ten-day average.

The requirements for Nicklaus By-Products (Table 8-6) were significantly more stringent as a result of the reverse flow-weighting approach used in the WLA. The WLA for toxicity was as follows: seven-day average less than 1.5 TUs and one-day value less than 4.3 TUs measured as chronic toxicity to fathead minnows. The 25 datum points generated during the evaluation of the discharge had a mean of 45.5 TUs and a CV of 0.93. As with Cibesi, the variance was assumed constant, and the probability that the WLA would be exceeded was set at 0.01. The LTA required to meet the seven-day WLA requirement was more limiting than the LTA needed to meet the one-day WLA.

The daily maximum limit was calculated as 1.9 TUs. The average daily limit turned out to be below the detection limit for a toxicity test. (The detection limit of toxicity test is 1.0 TUs since samples cannot be diluted any less than 100% sample.) Because the permittee could not meet an average limit set at the detection limit and still take advantage of the short term variability allowed under the daily maximum, the State set the average limit at 1.2 TUs. The value was derived considering the weekly monitoring requirement and assuming that three measurements would be below the detection limit (< 1.0 TUs) and one could be as high as 1.9 TUs. It was felt these limits best approximated the required effluent performance.

For cyanide, the one-day WLA value was 30 μg/l and the seven-day value was 6 μg/l. Since the State had no basis to estimate the CV, and the seven-day value was so close to the detection limit for cyanide (5 μg/l), the State did not use a statistical analysis. The treatment that the company proposed consisted of recycling; segregation of high strength wastestreams; polymer settling; alkaline chlorination; recombination of wastestreams; extended aeration-activated sludge with powered activated carbon; and clarification. It was felt that no cyanide would be discharged. The State set cyanide limits of 12 μg/l daily maximum and 6 μg/l average monthly with daily monitoring for the first year.

Appendices

A. Policy for the Development of Water Quality-based Permit Limitations for Toxic Pollutants

B. Sampling

C. Ambient Toxicity Testing and Data Analysis

D. Duration and Frequency

E. Lognormal Distribution and Permit Limit Derivations

Appendix A.
Policy for the Development of Water Quality-based Permit Limitations for Toxic Pollutants

[OW-FRL-2533-1]

Development of Water Quality-Based Permit Limitations for Toxic Pollutants; National Policy

AGENCY: Environmental Protection Agency (EPA).

ACTION: Notice.

SUMMARY: EPA has issued a national policy statement entitled "Policy for the Development of Water Quality-Based Permit Limitations for Toxic Pollutants." This policy addresses the technical approach for assessing and controlling the discharge of toxic substances to the Nation's waters through the National Pollutant Discharge Elimination System (NPDES) permit program.

FOR FURTHER INFORMATION CONTACT: Bruce Newton or Rick Brandes, Permits Division (EN-336), Office of Water Enforcement and Permits, U.S. Environmental Protection Agency, Washington, D.C. 20460, 426-7010.

SUPPLEMENTARY INFORMATION: As the water pollution control effort in the United States progresses and the "traditional" pollutants (oxygen demanding and eutrophying materials) become sufficiently treated to protect water quality, attention is shifting towards pollutants that impact water quality through toxic effects. Compared with the traditional pollutants, regulation of toxic pollutants is considerably more difficult. The difficulties include (1) the great number of toxic chemicals that may potentially be discharged to receiving waters and the difficulties in their analysis; (2) the changes in the toxic effects of a chemical resulting from reactions with the matrix of constituents in which it exists; and (3) the inability to predict the effects of exposure to combinations of chemicals.

To overcome some of these problems, EPA and the States have begun to use aquatic toxicity tests and various human health assessment techniques to complement chemical analyses of effluents and receiving water samples. Because these techniques or their application to effluent testing are new, EPA and the States have been cautious in their use. Based on EPA's evaluation of these techniques and the experiences of several States, EPA is now recommending the use of biological techniques as a complement to chemical-specific analyses to assess effluent discharges and express permit limitations. EPA has issued these recommendations through a statement of policy and is developing a technical guidance document to help implement the policy.

The complete test of the national policy statement follows:

Policy for the Development of Water Quality-Based Permit Limitations for Toxic Pollutants

Statement of policy

To control pollutants beyond Best Available Technology Economically Achievable (BAT), secondary treatment, and other Clean Water Act technology-based requirements in order to meet water quality standards, the Environmental Protection Agency (EPA) will use an integrated strategy consisting of both biological and chemical methods to address toxic and nonconventional pollutants from industrial and municipal sources. Where State standards contain numerical criteria for toxic pollutants, National Pollutant Discharge Elimination System (NPDES) permits will contain limits as necessary to assure compliance with these standards. In addition to enforcing specific numerical criteria, EPA and the States will use biological techniques and available data on chemical effects to assess toxicity impacts and human health hazards based on the general standard of "no toxic materials in toxic amounts."

EPA, in its oversight role, will work with States to ensure that these techniques are used wherever appropriate. Under section 308 and section 402 of the Clean Water Act (the Act), EPA or the State may require NPDES permit applicants to provide chemical, toxicity, and instream biological data necessary to assure compliance with standards. Data requirements may be determined on a case-by-case basis in consultation with the State and the discharger.

Where violations of water quality standards are identified or projected, the State will be expected to develop water quality-based effluent limits for inclusion in any issued permit. Where necessary, EPA will develop these limits in consultation with the State. Where there is a significant likelihood of toxic effects to biota in the receiving water, EPA and the States may impose permit limits on effluent toxicity and may require an NPDES permittee to conduct a toxicity reduction evaluation. Where toxic effects are present but there is a significant likelihood that compliance with technology-based requirements will sufficiently mitigate the effects, EPA and the States may require chemical and toxicity testing after installation of treatment and may reopen the permit to incorporate additional limitations if needed to meet water quality standards. (Toxicity data, which are considered "new information" in accordance with 40 CFR 122.62(a)(2), could constitute cause for permit modification where necessary.)

To carry out this policy, EPA Regional Administrators will assure that each Region has the capability to conduct water quality assessments using both biological and chemical methods and provide technical assistance to the States.

Background

The Clean Water Act establishes two principal bases for effluent limitations. First, existing dischargers are required to meet technology-based effluent limitations that reflect the best controls available considering economic impacts. New source dischargers must meet the best demonstrated technology-based controls. Second, where necessary, additional requirements are imposed to assure attainment and maintenance of water quality standards established by the States and approved by EPA. In establishing or reviewing NPDES permit limits, EPA must ensure that the limits will result in the attainment of water quality standards and protect designated water uses, including an adequate margin of safety.

For toxic and nonconventional pollutants it may be difficult in some situations to determine attainment or nonattainment of water quality standards and set appropriate limits because of complex chemical interactions which affect the fate and ultimate impact of toxic substances in the receiving water. In many cases, all potentially toxic pollutants cannot be identified by chemical methods. In such situations, it is more feasible to examine the whole effluent toxicity and instream impacts using biological methods rather than attempt to identify all toxic pollutants, determine the effects of each pollutant individually, and then attempt to assess their collective effect.

The scientific basis for using biological techniques has advanced significantly in recent years. There is now a general consensus that an evaluation of effluent toxicity, when adequately related to instream conditions, can provide a valid indication of receiving system impacts. This information can be useful in developing regulatory requirements to protect aquatic life, especially when data from toxicity testing are analyzed in conjunction with chemical and ecological data. Generic human health effects methods, such as the Ames mutagenicity test, and structure-activity relationship techniques are showing promise and should be used to identify potential hazards. However, pollutant-specific techniques are the best way to evaluate and control human health hazards at this time.

Biological testing of effluents is an important aspect of the water quality-based approach for controlling toxic pollutants. Effluent toxicity data in conjunction with other data can be used to establish control priorities, assess compliance with State water quality standards, and set permit limitations to achieve those standards. All States have water quality standards which include narrative statements prohibiting the discharge of toxic materials in toxic amounts. A few State standards have criteria more specific than narrative criteria (for example, numerical criteria for specific toxic pollutants or a toxicity criterion to achieve designated uses). In States where numerical criteria are not specified, a judgment by the regulatory authority is required to set quantitative water quality-based limits on chemicals and effluent toxicity to assure compliance with water quality standards.

Note.—Section 308 of the Act and corresponding State statutes authorize EPA and the States to require of the owner/operator any information reasonably required to determine permit limits and to determine compliance with standards or permit limits. Biological methods are specifically mentioned. Toxicity permit limits are authorized under Section 301 and 402 and supported by Section 101.

Application

This policy applies to EPA and the States. The policy addresses the use of chemical and biological methods for assuring that effluent discharges are regulated in accordance with Federal and State requirements. This policy was prepared, in part, in response to concerns raised by litigants to the Consolidated Permit Regulations (see FR 52079, November 18, 1982). Use of these methods for developing water quality standards and trend monitoring are discussed elsewhere (see 48 FR 51400, November 8, 1983 and *Basic Water Monitoring Program* EPA-440/9-76-025). This policy is part of EPA's water quality-based control program and does not supersede other regulations, policy, and guidance regarding use attainability, site-specific criteria modification, wasteload allocation, and water quality management.

Implementation

State Role

The control of toxic substances to protect water quality must be done in the context of the Federal-State partnership. EPA will work cooperatively with the States in identifying potential water quality standards violations, assembling relevant data, developing appropriate testing requirements, determining whether standards are being violated, and defining appropriate permit limits.

Note.—Under sections 303 and 401 of the Act, States are given primary responsibility for developing water quality standards and limits to meet those standards. EPA's role is to review the State standards and limits and develop revised or additional standards or limits as needed to meet the requirements of the Act.

Integration of Approaches

The type of testing that is most appropriate for assessing water quality impacts depends on the type of effluent and discharge situation. EPA recommends that an integrated approach, including both biological and chemical techniques, be used to assess and control water quality. The principal advantages of chemical-specific techniques are that (1) chemical analyses are usually less expensive than biological measurements in simple cases; (2) treatment systems are more easily designed to meet chemical requirements than toxicity requirements; and (3) human health hazards and bioaccumulative pollutants can best be addressed at this time by chemical-specific analysis. The principal advantages of biological techniques are that (1) the effects of complex discharges of many known and unknown constituents can be measured only by biological analyses; (2) bioavailability of pollutants after discharge is best measured by toxicity testing; and (3) pollutants for which there are inadequate chemical analytical methods or criteria can be addressed.

Pollutant-specific chemical analysis techniques should be used where discharges contain few, well-quantified pollutants and the interactions and effects of the pollutants are known. In addition, pollutant-specific techniques should be used where health hazards are a concern or bioaccumulation is suspected. Biological techniques should be used where effluents are complex or where the combined effects of multiple discharges are of concern. EPA recognizes that in many cases both types of analysis must be used.

Testing Requirements

Requirements for dischargers to collect information to assess attainment or nonattainment of State water quality standards will be imposed only in selected cases where the potential for nonattainment of water quality standards exists. Where water quality problems are suspected but there is a strong indication that complying with BCT/BAT will sufficiently mitigate the impacts, EPA recommends that applicable permits include testing requirements effective after BCT/BAT compliance and reopener clauses allowing reevaluation of the discharge.

The chemical, physical, and biological testing to be conducted by individual dischargers should be determined on a case-by-case basis. In making this determination, many factors must be considered, including the degree of impact, the complexity and variability of the discharge, the water body type and hydrology, the potential for human health impact, the amount of existing data, the level of certainty desired in the water quality assessment, other sources of pollutants, and the ecology of the receiving water. The specific data needed to measure the effect that a discharger has on the receiving water will vary according to these and other factors.

An assessment of water quality should, to the extent practicable, include other point and nonpoint sources of pollutants if the sources may be contributing to the impacts. Special attention should be focused on Publicly Owned Treatment Works (POTW's) with a significant contribution of industrial waste-water. Recent studies have indicated that such POTW's are often significant sources of toxic materials. When developing monitoring requirements, interpreting data, and determining limitations, permit engineers should work closely with water quality staff at both the State and Federal levels.

A discharger may be required to provide data upon request under section 308 of the Act, or such a requirement may be included in its NPDES permit. The development of a final assessment may require several iterations of data collection. Where potential problems are identified, EPA or the State may require monitoring to determine whether more information is needed concerning water quality effects.

Use of Data

Chemical, physical, and biological data will be used to determine whether, after compliance with BCT/BAT requirements, there will be violations of State water quality standards resulting from the discharge(s). The narrative prohibition of toxic materials in toxic amounts contained in all State standards is the basis for this determination taking into account the designated use for the receiving water. For example, discharges to waters classified for propagation of cold water fish should be evaluated in relation to acute and chronic effects on cold water organisms, potential spawning areas, and effluent dispersion.

Setting Permit Limitations

Where violations of water quality standards exist or are projected, the State and EPA will determine pollution control requirements that will attain the receiving water designated use. Where effluent toxicity is an appropriate control parameter, permit limits on effluent toxicity should be developed. In such cases, EPA may also require a permittee to conduct a toxicity reduction evaluation. A toxicity reduction evaluation is an investigation conducted within a plant or municipal system to isolate the sources of effluent toxicity, specific causative pollutants if possible, and determine the effectiveness of pollution control options in reducing the effluent toxicity. If specific chemicals are identified as the cause of the water quality standards violation, these individual pollutants should be limited. If a toxicity reduction evaluation demonstrates that limiting an indicator parameter will ensure attainment of the water quality-based effluent toxicity requirement, limits on the indicator parameter should be considered in lieu of limits on effluent toxicity. Such indicator limits are not limits on causative pollutants but limits demonstrated to result in a specific toxicity reduction.

Monitoring

Where pollution control requirements are expressed in terms of a chemical or toxicological parameter, compliance monitoring must include monitoring for that parameter. If an indicator parameter is used based on the results of a toxicity reduction evaluation, periodic toxicity testing may be required to confirm the adequacy of the indicator. Where biological data were used to develop a water quality assessment or where the potential for water quality standards violations exist, biological monitoring (including instream monitoring) may be required to ensure continuing compliance with water quality standards.

EPA believes that the intelligent application of an integrated strategy using both biological and chemical techniques for water quality assessment will facilitate the development of appropriate controls and the attainment of water quality standards. EPA looks forward to working with the States in a spirit of cooperation to further refine these techniques.

Policy signed February 3, 1984 by Jack E. Ravan, Assistant administrator for Water.

Dated: February 16, 1984.

Jack E. Ravan,

Assistant Administrator for Water.

[FR Doc. 84-6445 Filed 3-8-84; 8:45 am]

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Appendix B.
Sampling

Sampling

The objective of an effluent or instream sampling program is to obtain a sample (or samples) from which a representative measure of the parameter of interest can be obtained. Unfortunately, many of the industrial and municipal NPDES sampling protocols presently in use are carryovers from schemes used for calculating loadings of nutrients and oxygen-demanding substances, or were developed to evaluate treatment plant operational efficiency. Sampling for individual toxicants and particularly for effluent toxicity can require more specific (and thus different) sampling procedures.

Wastewater variability is an important consideration in selecting the method and frequency of sampling for both chemical analysis and toxicity testing. Industrial waste characteristics have been shown to vary in frequency, intensity, and duration (1). As noted by Bender (2), the sources of effluent variability include both random and systematic components that influence both daily and annual characteristics of waste discharges. Although toxic pollutant loading may be of primary concern in assessing human health impact or bioaccumulation, loading may be of lesser importance in toxicity assessment than frequency, intensity, and duration of peak toxic discharge. Sampling must be tailored to measure the type of toxicity of importance for that discharge: either long term (chronic) impact, which is a more constant effect, or short term (acute) impact, which is more variable and subject to peaks of intensity.

There are several chemical parameters for which continuous analysis is possible. These include pH, temperature, dissolved oxygen, and other parameters involving instantaneous measurement. All other types of measurement involve some time period over which the analysis is conducted. Toxicity tests require an exposure period. Chemical tests require sample preparation and analysis. There is no continuous analysis method for toxicity.

It should be noted that although it is difficult to design a representative sampling program for toxicity analysis, the problems are of no greater magnitude than similar problems associated with obtaining a representative sample for conventional pollutants.

Sampling Methods

Continuous Flow Samples

For toxicity testing, the test organisms may be exposed to serial dilutions of a sample continuously pumped from the effluent pipe or ditch. In the case of effluents, if optimum accuracy is desired, then the ratio of effluent flow/test chamber volume can be scaled to simulate the time-varying concentration at the mixing zone boundary.

Although flow-through methods can provide a realistic simulation of time-varying exposure, they are relatively expensive and are usually conducted on site. Therefore, flow-through methods may only be practical where the goals of the analysis of impact require this type of testing or where treatment costs are sufficiently high that this type of analysis can be required. A flow-through exposure method is not a continuous analysis because only one result or data point is obtained at the end of the test. However, the continuous exposure does provide some measure of time-varying exposure effects.

Discrete Samples

Grab or flow composited sampling provide a discrete sample for chemical analysis or toxicity testing. Static or renewal toxicity tests using discrete samples result in exposure of test organisms to a constant effluent composition over the period of the tests, or for the period between renewals.

If discrete samples are collected during peaks of effluent toxicity, then constant concentration exposure static tests provide a measure of maximum effect.

Depending on the duration of a peak and the compositing period, composited samples may not be useful for examining toxicity peaks because the compositing process tends to dilute the peaks. Composited samples are usually appropriate for chronic tests where peak toxicity of short duration is of less concern. The averaging effect of compositing may be misleading when testing for acute toxicity.

Grab samples must be collected at sufficiently frequent intervals to provide a high probability of sampling daily peaks. Fortunately, static toxicity tests are relatively inexpensive and can be done on shipped samples; thus, it may be cost effective to conduct individual tests on a series of grab samples collected over a 24-hour period.

Sampling Frequency

Non-random effluent variability, resulting from batch processing, variable loadings, etc., is often known or can be determined. Therefore, the first step in designing a sampling program for chemical analysis or toxicity testing is to select the annual sampling frequency based on available site-specific operational information. This is important in selecting sampling periods for both continuous flow and discrete sampling methods.

If discrete sampling methods (grabs or composites) are used, then random variations between and within days for each sampling period must be considered. It is important to recognize the tradeoff between the long term (between days) frequency and short term (within days) frequency of sample collection and analysis for toxics. At present, the permit requirements for sampling and analyzing chemical parameters are site specific and generally involve a single grab or 24-hour composite sample collected at daily, weekly, or monthly intervals. Unfortunately, a sampling scheme involving a single daily grab or a 24-hour composite sample can conceal the presence of those daily extreme values that may be of importance. To optimize sampling cost and effectiveness, it may be desirable to reduce long term frequency so that daily frequency can be increased.

For example, a weekly grab or composite involves 52 analyses per year. It may be more efficient to reduce the annual frequency to monthly or bimonthly, but collect and analyze four or eight grabs daily. Either scheme (12 × 4 or 6 × 8) would involve 48 analyses per year vs. 52 for the weekly single sample approach. Assuming that daily toxic events of environmentally significant intensity and duration would not be masked by short

Box B-1
RECOMMENDATIONS

The initial sampling design step should involve stratification of sampling periods to account for non-random sources of variation (e.g., batch processing). The second step includes selection of the frequency and the method of sampling to be conducted within each sampling period. Depending on site-specific considerations, several options are available.

- *Flow-through Methods* — Ideally, for both acute and chronic effluent toxicity tests, the exposure of biota should simulate the time-varying concentration at a predetermined point in the receiving water. For regulatory purposes, the critical point is often the edge of the mixing zone where the waste should exhibit neither acute nor chronic toxicity. Therefore, if warranted by site-specific factors, it is recommended that test biota be exposed to a continuously collected flowthrough sample of serially diluted effluent. If no systematic annual variations (e.g., batch processing) are known or suspected, flow-through testing can be conducted at a minimum of quarterly intervals for at least one year.
- *Grab Sample Methods* — Grab samples are recommended for chemical analyses and for acute and chronic toxicity tests where site conditions (such as wastewaters that are known to have relatively constant composition) do not require use of continuous flow methods. Grab samples of effluent or receiving water may be used for static or renewal acute toxicity tests, which may be conducted on site or at a remote lab. The design of a toxics grab-sampling program must take into account the trade-off between long term and short term sampling intensity. Where there is no ponding of wastes or retention time is insufficient for thorough mixing, it is important to collect or analyze a sufficient number of samples to provide a measure of daily spikes. Therefore, to minimize analytical costs where daily fluctuations are known or suspected, the annual sampling frequency should be reduced in favor of more intensive daily sampling. It is recommended that on an annual cycle, grab sampling and analysis include a minimum of four to six daily grabs collected monthly. An option could include the use of short term (four-hour) composites rather than grabs. If site-specific data are available to indicate that treatment system retention time is adequate to minimize daily variations, then the daily replicates may be omitted in favor of more frequent annual sampling (e.g., weekly or semimonthly rather than monthly). If, to minimize costs in screening tests, only single samples are collected at infrequent intervals (e.g., quarterly) an uncertainty factor for variability should be used in the toxicity evaluation (see Section 3).
- *Composite Sample Methods* — If static or renewal methods are used for evaluation of toxicity, it is recommended that 24-hour, continuous-flow composite samples be collected. Considerations of annual frequency are the same as those for grab samples.

term composites, it might be more efficient to collect eight samples each composited over a three-hour interval.

If costs or other constraints prohibit satisfactory daily and annual replication of sampling, then a level of uncertainty must be introduced into the calculations used to evaluate waste toxicity (see Section 3, Table 3-1).

Box B-1 presents EPA's recommendations on sampling methods.

References

1. Nemetz, P.N., and H.D. Dreschler. 1978. The role of effluent monitoring in environmental control. J. Water, Air, and Pollution. 10:477-497.
2. Bender, E.S. 1984. Sources of variations in effluent toxicity tests. In: Environmental Hazard Assessment of Effluents. (H. Bergman, R. Kimerle, and A.W. Maki, eds.) Proceedings of the Fifth Pellston Environmental Workshop. Cody, WY, August, 1982.

Appendix C.
Ambient Toxicity Testing and Data Analysis

Ambient Toxicity Analysis

Ambient toxicity testing procedures are useful where measurement of toxicity levels after discharge is important in the assessment of toxic effluent impact. This is particularly true where impact is caused by the presence of multiple point sources. The purpose of this testing is to provide an analysis of toxicity levels instream from whatever sources of toxicity are affecting the receiving water.

Procedures

The basic ambient toxicity testing procedure is to expose test organisms to receiving water samples taken from selected sampling stations above, at, and below the discharge point(s). Since effluent concentrations after discharge are often relatively low, chronic toxicity tests should be conducted so that the tests are sensitive enough for the purpose.

The methods available for chronic testing of sufficiently short duration are limited. Two organisms for which short term chronic toxicity tests are available are *Pimephales promelas* and *Ceriodaphnia sp.*

The following procedures are used:

- Select instream sampling stations based on the mixing characteristics involved in the specific discharge situation.
- Collect a daily grab sample or a daily composite sample of receiving water from each station.
- Use a renewal testing method to expose test organisms to the daily samples collected at each station. Use an appropriate number of replicates (10 for *Ceriodaphnia*) for each sampling station. No dilution series is required where screening is the primary goal.
- Testing must be conducted at a low flow period. However, it is not necessary to conduct the tests at the critical low flow period. Testing is best when relatively stable flow occurs during the test period.
- Record the results of the testing in the format shown in Table C-1. The survival of the test organisms and the effect on their growth or reproduction are used as endpoints. Figure C-1 plots the results in graphic form so that the pattern of ambient toxicity can be observed.

Selecting Sampling Stations

The selection of sampling stations is determined by the characteristics of the site. When determining stations, the following factors should be considered:

- Mixing and flow — The mixing characteristics of the discharge site are useful to determine the placement of sampling stations. Knowledge of concentration isopleths allows the regulatory authority to place stations at locations instream

Table C-1. Young Production and Percent Survival of Ceriodaphnia in Ambient Toxicity Tests at Ottawa River, Lima, Ohio

Station	Station Description	River Mile	Young Female	SD	Final Survival	Daily Survival 1	2	3	4	5	6	7
1	Above Lima	46.0	15.5	8.0	90	100	100	100	90	90	90	90
2	Above STP	37.7	14.1	2.1	0	100	100	100	100	90	10	0
3	Below STP	37.4	0	—	0	100	100	10	0	0	0	0
3A	Midway between STP and refinery	37.3	0	—	0	100	100	10	0	0	0	0
3B	Above refinery	37.1	0.4	—	0	90	90	40	0	0	0	0
4	Above chemical plant	36.9	7.5	3.6	10	100	100	100	100	100	50	10
4A	Below chemical plant	36.3	11.1	4.6	30	100	100	100	100	100	40	30
5	Shawnee Bridge	35.4	5.7	4.0	0	90	90	90	90	90	60	0
6	Route 117	32.6	12.6	3.8	10	100	100	100	100	100	100	10
7	Allentown	28.8	16.8	6.1	100	100	100	100	100	100	100	100
8	Rimer	16.0	17.4	9.5	80	100	90	90	90	90	80	80
8A	"Boonie" Station	8.0	25.0	3.3	100	100	100	100	100	100	100	100
9	Kalida	1.0	25.6	5.5	100	100	100	100	100	100	100	100

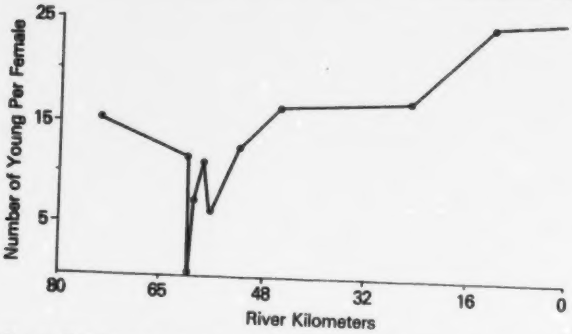

Figure C-1. Ceriodaphnia young production in water from various stream stations on the Ottawa River, Lima, Ohio.

that correspond to concentrations measured in the dilution series in the effluent tests. For example, where effluent testing shows the effluent NOEL is 10%, an instream station should be placed where dilution is estimated to create a 10% instream waste concentration. In this way, the size of a toxic plume can be measured. Sampling stations should be placed where the effluents exist at relatively constant and relatively specific concentrations. Test at specific low flow conditions, if possible. Presence of tributaries or other sources of dilution will influence positions and numbers of stations. Where smaller tributaries have several point sources on them, treat the tributary as a point source. Obvious non-point source areas should also be used to set stations.

- Existing biological data — Where biosurvey data are available, sampling station location should be influenced by the more obvious trends in impact. In particular, control stations and recovery stations can be determined by biosurvey data.
- Single point sources — Single point source situations should be bracketed with an above station, an immediate mixing station, several intermediate stations corresponding to different instream concentrations, and a recovery station. Of course, a control station should be established.
- Presence of other point sources — Multiple point source situations require the placement of more stations between discharge points. Each source should be bracketed by sampling stations.

There are four areas or zones that can be recognized when establishing the sampling stations for ambient toxicity testing:

- Zone 1 — An upstream zone before the effluent enters.
- Zone 2 — A zone of mixing.
- Zone 3 — A zone after mixing and before additional dilution water enters.
- Zone 4 — A zone where additional dilution occurs either from effluents or tributaries.

All possible combinations of occurrences are not practical to discuss but must be sorted out for each site. Some generalizations are important to mention:

- Any upstream sources of contaminants, such as other discharges, will confound the individual effects of a downstream discharge. For example, Zone 3 of the downstream discharge may occur in Zone 4 of an upstream discharge. This does not invalidate the measurement of ambient toxicity. It only makes it difficult to attribute amounts of response to each individual discharge. Response to the instream mixture is what is measured.
- Careful location of sampling stations in Zone 3 is critical. Zone 3 is the only place where toxicity decay rates of any one discharger can be measured and then only if there are no upstream discharges, or if there are, only if that upstream effluent is stable in that reach.
- In Zone 4, not only is degradation of the effluent toxicity occurring, but there is dilution of it by other effluents and tributaries. Depending on the site circumstances, one may not be able to learn anything about the ambient toxicity characteristics of the effluent of concern in this zone.
- To emphasize, what can be measured in each zone depends on the above considerations. In the more complex situation, only an estimate of ambient toxicity at each station can be obtained. No information about one effluent's toxicity decay rate will be available where several toxic effluents mix. In the most simple situation of one discharge and no dilution downstream for a long distance, Zone 3 will be large enough to get a good measure of toxicity decay.

Analysis of Ambient Toxicity Measurement

- When used in screening, the ambient toxicity data can identify areas in receiving waters where ambient toxicity exists instream. Attributing such impact to specific point sources (particularly where several sources discharge) may require effluent toxicity testing.
- Except when used for screening purposes, ambient toxicity measurements must be interpreted in conjunction with effluent toxicity test data if conclusions are to be drawn concerning changes in toxic effect after discharge. The same species must be used in both the ambient and the effluent toxicity tests.
- When analyzing the data, the performance of the animals at each station downstream is compared to that of the animals exposed to receiving water without the effluent of concern in it but containing all other upstream additions. The result is an integration of effects from all contaminants and components and represents not only the toxicity of the effluent of concern but also the interactions of it with other effluents.

- Where the downstream stations show toxic effect at the concentrations measured as toxic in the effluent toxicity tests, effluent toxicity can be considered to be occurring instream, after discharge.
- Where the toxic effect decreases from station to station downstream in the absence of further dilution, the effluent toxicity is degrading. If the decay rate is rapid (e.g., no toxicity at the closest instream station to the discharge point), the effluent has a non-persistent toxicity. Where the decay rate is more gradual, toxicity is more persistent. The rate of decay of toxicity together with mixing data allows the regulatory authority to approximate a receiving water toxicity impact area. That impact area can then be compared to the appropriate state water quality standards when establishing control requirements.
- In some cases, ambient toxicity may increase in relation to effluent toxicity measurements. Either upstream sources of toxicity exist or some factor in the receiving water is reacting with the effluent to increase its toxicity. Again, the pattern and magnitude of change in toxicity should be analyzed. Differences in toxicity levels between stations will reveal what is happening to the effluent as it is mixed instream and interacts with the constituents of the receiving water.
- Trend analysis in the raw test data is of importance in the interpretation of ambient toxicity data. As used in this context, trend analysis means observing toxic effect as it occurs in the test itself and relating it to what is occurring instream (plug flow, intermittent discharge, peak toxicity of effluents). Using time-of-travel data or receiving water flow rates and patterns, observe effects on the test organisms from day to day. There may be a pattern of mortality which can be linked to discharge events. For example, in C-1 the data indicate late mortality at downstream stations on days 6 and 7. Flow rates for the river in this example correlated this mortality to the downstream movement of a toxic slug illegally discharged upstream.

Appendix D.
Duration and Frequency

The following discussion describes the EPA's recommendations for duration and frequency as they apply to the assessment and control of toxic pollutants. It should be noted that the recommendations presented are based directly on national water quality criteria. They are subject to change if and when EPA determines any changes, improvements, or updates are needed.

Two important factors must be accounted for when assessing exposure conditions and setting water quality-based permit limits for toxicants in NPDES permits: duration of adverse exposure and frequency with which criteria are exceeded. Specifications for these factors are provided in the national water quality criteria documents. They can be defined as follows:

- Duration of exposure considers the amount of time organisms will be exposed to toxicants. It is expressed as that period of time over which the instream concentration is averaged for comparison with criteria concentrations. The duration of exposure is determined scientifically using primarily laboratory data.
- Frequency is defined as how often exposures that exceed the criteria can occur during a given period of time (e.g., once every ten years) without unacceptably affecting the community. Ideally, a dynamic model of the exposed population would be used to select the frequency. Currently this modeling is beyond our capabilities, so the choice of the frequency represents the best judgement of how often a population can be reduced without affecting its viability.

Duration

Several factors must be considered in order to properly control duration. Toxic effect is a function of both magnitude (concentration) and duration (time). Toxicity tests measure concentrations that cause adverse impacts over the testing period. In addition to limiting the concentration of toxicants, the regulatory agency must also specify and limit the length of time the instream biota are exposed to those concentrations. For example, a very brief exposure to a relatively high concentration may be less harmful than a prolonged exposure to a lower concentration. Therefore, two duration specifications need to be set: a maximum duration for protection against acute effects, and a maximum duration for protection against chronic effects.

For acute effects, the duration must be as short as practicable because acute toxicity can occur quickly. An instantaneous value as a maximum limit would be most protective. Determination of an instantaneous maximum would require impractical and resource-intensive continuous monitoring. Therefore, *acute criteria are expressed as a criterion maximum concentration (CMC) occuring in a one-hour averaging period.* The hourly average is based on a daily measurement in most cases.

It is more complicated to set the tolerable duration of exposure for chronic effects. Since chronic toxicity occurs over an extended time period, chronic effect concentrations can be exceeded for a relatively short period of time with no adverse effect so long as the CMC is not exceeded. Chronic criteria are expressed as a criterion continuous concentration (CCC). The objective is to maintain the concentration of a toxicant (or toxicity) at or below the CCC over a protracted period of time, but still allow some excursion above the CCC that will not cause toxic effect. The question is, how long can concentrations exceed the CCC without some adverse impact occurring?

Exposure concentrations are usually continuously fluctuating. Figure D-I graphically represents how concentrations could vary over time. Accurate measurement of the fluctuations over time would require continuous monitoring. As stated above, most measurements are

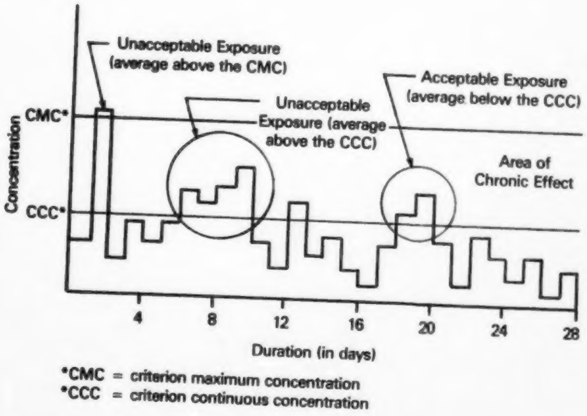

Figure D-1. Idealized representation of duration of acceptable and unacceptable toxic concentrations.

taken on a daily basis and consist of either instantaneous measurements or some form of composite measurement. Water quality analysts also typically examine environmental parameters as an average over some time period. Because concentrations can be above the CCC without causing adverse effects, there is considerable temptation to specify the CCC in terms of average exposure. However, if the period during which exposure is averaged is long, periods of concentrations above the CCC can produce unacceptable toxic effects without the average concentration exceeding the CCC.

CCCs are usually based on test concentrations that are constant. Therefore, they do not provide data for effects of exposures fluctuating about an average, even though this is the type of exposure that occurs in receiving waters.

The following examples illustrate what happens when organisms are exposed to toxic effluents at specific concentrations known to produce acute and chronic toxic effects. These four examples are taken from the Complex Effluent Toxicity Testing Program (a multi-year EPA research effort examining the toxicity of industrial and municipal effluents). The examples show situations where substantial toxicity to the test species *Ceriodaphnia affinis* results from exposure at concentrations below the CMC and above the CCC for periods of time from five to seven days.

Lima STP

LC50 = 100%

Dilution required to meet the CMC
(LC50×0.3) = 30%

NOEL = 5.5%

At a 30% effluent concentration, 40% of the *Ceriodaphnia* died during days 6 and 7. At a 10% concentration, there were no deaths after four days. On days 5 and 7, however, young production was 24% and 30% less than the production resulting from exposure to a 3% effluent concentration. These data show that exposure concentrations above the CCC (NOEL) but below the CMC (0.3 ×LC50) can definitely be harmful.

Petroleum Refinery

LC50 = >100%

Dilution required to meet the CMC
(LC50×0.3) = >30%

NOEL = 5.5%

There were no *Ceriodaphnia* fatalities up through seven days' exposure at a 30% effluent concentration, but there was zero young production during that period. At a 10% concentration, there were no mortalities. However, at 5 and 7 days, young production was 43% and 40% less than the young production resulting from exposure to a 3% effluent concentration.

Birmingham Coke Plant

LC50 = >100%

Dilution required for the CMC
(LC50×0.3) = >30%

NOEL = 5.5%

At a 30% effluent concentration, there were no *Ceriodaphnia* deaths, but zero young production for all days. At a 10% effluent concentration, there were no mortalities. On days 4, 5, 6, and 7, young production was reduced 57%, 94%, 98%, and 86%, respectively, as compared to young production resulting from exposure to a 3% effluent concentration.

Naugatuck STP

LC50 = >100%

Dilution required for the CMC
(LC50×0.3) >30%

NOEL = 17%

At a 30% effluent concentration, 10% of the *Ceriodaphnia* died after the 4th day, and young production was reduced on days 5 and 7 by 24% and 62% as compared to production resulting from exposure to a 10% effluent concentration.

No similar data for fathead minnows (also tested) can be given because the only measurements made on days 5, 6, and 7 are for lethality and the CMC (0.3 times the LC50) is supposed to eliminate lethality. Based on data for effluents currently available, the CMC as now set seems to prevent lethality. In order to gather similar data for fatheads, exposures would have to be terminated on days 5 and 6 to get the weights. This has not been done.

These data combined with other observations taken during chronic toxicity testing show that four days is the maximum period during which concentrations may be continuously above the CCC (it must be remembered that the CMC should not be exceeded for more than one hour). After four days, impact begins to occur. Since the longest duration allowable above the CCC must be limited to four days or less, in the majority of discharge situations *the maximum period in which the average exposure should not exceed the CCC is 4 days.*

Exposure to substantial fluctuations about a mean concentration causes more adverse effect than continuous exposure to the mean concentration. The averaging periods recommended to restrict duration of adverse exposure have been made short enough to restrict allowable fluctuation in the instream concentrations. The CCC stream flow averaging period used for steady state wasteload allocation (WLA) modeling may be as long as 30 days in situations involving publicly owned treatment works (POTWs) designed to remove ammonia where limited variability of effluent pollutant concentration and resultant concentrations in receiving waters can be demonstrated. In cases where low variability can be demonstrated, longer averaging periods for the ammonia CCC (e.g., 30-day averaging periods) would be acceptable because the magnitude and duration of concentrations above the CCC would be sufficiently limited.

Frequency

Developing a WLA involves assessing the probability that an event (criteria being exceeded) will occur. To be completely certain that an event will never happen would require unnecessarily stringent pollution controls. Ecosystems can tolerate occasional impacts provided they are not too severe or too frequent. The question is how often criteria can be exceeded without adversely affecting the community. Exceeding criteria here refers to either a one-hour average exceeding the CMC or a four-day average exceeding the CCC (Figure D-l).

The guidance for how frequently criteria should be allowed to be exceeded is developed along two lines. These are based on the severity of the expected biological impact and are discussed below.

Considerations for Setting Frequency for Minor Stresses

Criteria may be exceeded either as a result of predictable impacts from regulated pollutants or because of spills (which are not regulated discharges under permit limits but can have drastic effects on the community). In the Complex Effluent Toxicity Testing Program, eight field studies evaluating the use of toxicity tests to predict biological impact were conducted, and ambient toxicity measurements were taken over a seven-day period. During three of these studies, spills of pollutants resulting in acute toxicity were documented. This seemingly high frequency of spills suggests that their effects may be as important as impacts caused by normal variations in effluent composition, concentration, and dilution flow.

Predictable impacts, the occurrence of low streamflow and high effluent concentration or toxicity, can be reasonably well modeled and incorporated into the permit. These impacts should most often be minor. Special care should be exercised when there are many outfalls in a small reach of receiving water, however, because if one discharge exceeds the criteria due to low flow, criteria violations are very likely to occur at other outfalls at the same time. The several "minor" events may thus add up to a "major" one.

Communities that are not under stress can withstand an event better than a community that is under stress. Recovery time from small events in unstressed systems has been found to be a few months to a year (Table D-1). Logically then, the allowable frequency can be greater for unstressed communities.

In recovery studies where there was no toxicant residual present, most fish species could repopulate in as few as three weeks if conditions were advantageous. The recovery time depended on the season of the year. Recovery is most rapid in late summer to fall.

When determining the frequency that criteria may be exceeded, the extent of the area affected should also be considered. A reach of river having an affected area of a few hundred feet will probably recover in a few weeks, whereas a reach of several miles may take years. Recovery of lakes or isolated systems is believed to be very slow.

The considerations given to frequency should include the number of events that are expected in an individual year, the number of event-free years, and the number of years between events. The temporal distribution of events is dependent upon stream flow, effluent variability, and stream chemistry variability. Therefore, it is difficult to predict when and exactly how many events will occur. Most biological communities would not be sufficiently affected if on the average there was one event every three years. This means that there should be two years on the average between events. If multiple or very long events occur in one year, if uncontrolled spills occur, or if there are large areas of impact in relation to the size of the waterbody, there should be a longer period until the next event. It is possible in very select situations that a frequency of once every one or two years on the average might be acceptable. For such cases, it would have to be demonstrated that the affected area was small and the potential for biological recovery was high.

Considerations for Setting Frequency for Areas under Major Stresses and Sensitive Communities

The frequency with which criteria can be allowed to be exceeded will depend on the structure and function of the community and the spatial relationships to other non-affected areas. The impact on two different communities, given the same toxic effects, will depend on the availability of other unexposed receiving waters from which animals can repopulate an area affected by an exposure. If the sensitive individuals killed are long-lived and produce few young, the impact on the community will be greater than if short-lived organisms with higher reproduction potential are affected.

It is now believed that some ecosystems do not have the ability to recover in one human life span. Ecosystems can be classified as to their ability to recover after a severe stress over a large area as follows:

Ecosystem Type	*Ability to Recover*
Oceans	Low
Lakes	Slight
Rivers	Moderate
Estuaries	High

Cairns et al. [1] considered the following factors to be important in evaluating ecosystem recovery:

- Existence of nearby epicenters.
- Transportation or mobility of dissemules (eggs, spores).
- Condition of habitat following pollution stress.
- Presence of residual toxicants.
- Chemical and physical environment quality.
- Age and stability of ecosystem.

Characteristics of ecosystems that can be restored in structure and function are:

- Indigenous species accustomed to highly variable environmental conditions.
- System has high structural and functional redundancy.
- Low stream order, flow dependable, good turbulent diffusive action, high flushing.
- High water buffering.
- Not close to ecological Transition Zone.

Based on the above reference [1], ecosystem recovery of the Clinch and Roanoke Rivers was evaluated and approximations of species recovery were developed.

Years After Severe Major Stress	*Species Reestablished*
1st	40–60%
2nd	60–80%
3rd	80–95%

In the case of the Clinch River, the large clams had not recovered 13 years after the initial stress. It is probable in this system as well as in other reported cases that residual toxicity may have delayed the recovery.

Another important consideration is that a population that is recovering from a severe stress is similar in nature to a young system that is more vulnerable and cannot recover as fast, i.e. repeated events cause ever increasing impacts. The value placed on resources, especially as to "important species," will also be a factor in setting the frequency of allowed events.

The time required for various types of ecosystems to recover from a stressful event should be a primary consideration in determining the allowed frequency. Although there is not enough known to draw definitive conclusions relative to system type, some relative comparisons can be made with existing information. Much of this information is drawn from studies conducted subsequent to major oil spills and therefore reflects a bias toward near-shore saltwater systems affected by complex mixtures. Some work has been done on the recovery of freshwater systems from toxic spills. Table D-2 summarizes the relative susceptibility of various

Table D-1. Biological Recovery Studies

Site	Toxicant	Degree of Damage	Community Effect	Recovery Time	Reference
Clinch River Va.-Tenn.	Caustic water; Fly ash pond	Severe	Benthos 100%, fish 100%	After 2 years, not fully recovered	Cairns, et al. 1971
Smith Branch Illinois	Drought	Severe	Fish	At 3 weeks, 21 of 29 species recovered; Red Ear, 2 years later	Larimore, Childers and Heckraffe. 1959
Six sites on 4 streams in Louisiana	Electro-fishing	Decimation	Fish	4 of 6 sites in one year; other sites repopulated by other species	Berra, Gunning. 1970
Latort Spring Run Pennsylvania	Pesticide	Severe	Benthos 98%, fish	1 year at one of the sites; two years for complete recovery	Schott. 1981 Schott. 1982 Schott. 1983
Pine River Pennsylvania	Oil spill	Severe	Benthos	In 4 years 50% of species recovered; residual oil	Frey. 1973 Frey. 1974 Frey. 1975
Codorus Creek Pennsylvania	Pesticide	Severe	Benthos 80%, fish 90%	Good recovery in 2 months; continuing good recovery at 6 months	Schott. 1980
Cold Brook New Jersey	Oil	Moderate	Benthos, fish (trout)	2 years for recovery of y-o-y trout, 1 year for benthos	Collier. 1981
Straight River Minnesota	Zinc	Slight	Benthos, fish, zooplankton	At 6 weeks, recovering	Carlson. 1984

aquatic systems to initial impact and estimates time required for recovery following major stress.

Based on the above recovery data, an ecosystem capable of reestablishment would be back to its normal function in three to four years after a major stress. Based on logic, one would not want an ecosystem in a constant state of recovery and increased vulnerability from multiple stresses. An ecosystem severely stressed every few years will not reach its full potential.

Where the communities are under stress from 1) uncontrolled spills, 2) large areas of impact relative to the size of the water body (e.g., because of multiple dischargers), or 3) exposure that will be extended for longer time periods, the frequency should be reduced to once every four or more years on the average.

Recommendations for Duration and Frequency

The recommendations listed below are toxicologically based specifications to be used as inputs to WLA models. Depending on the modeling approach, different inputs may have to be used to ensure full attainment of these biological criteria. It is possible that a variety of WLA design flows could result in the attainment of the criteria. The following are the toxicologically based recommendations for ambient concentrations.

- The one-hour average concentration should not exceed the CMC more than once every three* years on the average.
- The four-day average concentration should not exceed the CCC more than once every three* years on the average.

The limitations of duration and frequency apply to both single chemical criteria and to effluent toxicity. The terms used are:

- For single chemicals the one-hour average concentration is equal to one-half the Final Acute Value and for general toxicity it is one-third the LC50 using the most sensitive of at least three test species.
- For single chemicals the four-day average concentration is equal to the lowest of the Final Chronic

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* If the biological community is under stress because of spills, multiple discharges, etc., or has a low recovery potential, or if a local species is very important, the frequency should be decreased.

Table D-2. Relative Susceptibility of Various Aquatic Systems to Initial Impact

Ecosystem Type	Time Between Major Stresses (in years)				
	3	5	10	20	100
Rivers					
Headwaters	50-70% of species recovered	Recovery less than 95% of species	Recovered	Recovered	Recovered
Middle Reach	50-75% of species recovered	State of constant recovery less than 95% of species	Recovered	Recovered	Recovered
Slow	50-75% of species recovered	Recovery less than 95% of species	Recovered	Recovered	Recovered
Lakes	Most species would not be recovered	Biological integrity not maintained	State of constant recovery	Final state of recovery	Recovered
Estuaries	Principally clams and molluscs are recovered	Clam and mollusc populations still reduced	Recovered	Recovered	Recovered
Marine					
Beaches	Beaches are in state of final repopulation	Repopulated and probably recovered	Recovered	Recovered	Recovered
Rock Shore	Colony communities not recovered	Colony communities generally recovered	Recovered	Recovered	Recovered
Tidal Flat	Principally bivalves not recovered	Bivalves still reduced	Recovered	Recovered	Recovered
Marshes	Annual plants and short life span organisms recovered	Long-lived plants not reestablished; most organisms recovered	Final stages of recovery	Recovered for very large systems	Recovery depends upon size of area affected
Open Water	Very small area repopulated	Long life span organisms in recovery	Most species present	Recovered except for very large systems	Recovery depends upon size of area affected

*For those capable of recovery

Recovered is defined as greater than 95 percent of the original species being present and those species capable of being present would be present.

Value, the Final Plant Value, and the Final Residue Value unless data show that a lower value should be used (National Guidelines), and for general toxicity it is the NOEL using the most sensitive of at least three test species.

The use of national or site-specific criteria in calculating a WLA for waste treatment facilities requires selection of either a steady state or a dynamic model. The concentrations, averaging periods, and frequencies with which the CCC and CMC criteria are allowed to be exceeded can be used directly in a dynamic model. Such models are preferable in that they give a more accurate picture of the stress on the environment by allowing prediction of the number and extent of events. However, limited data or other factors may make the use of this type of model impractical. In these cases, the regulatory authority may need to rely on a steady state model.

The use of steady state models requires selection of a design flow such that the biological criterion is not exceeded more often than allowed by the specified frequency. The allowed frequencies and the durations of the averaging periods specified in the criterion should not be used directly to calculate steady state design flows using extreme value analysis. For example, if a criterion specifies that the four-day average concentration should not exceed a particular value more than once every three years on the average, this should not be interpreted as implying that 4Q3 is appropriate to use as the design flow. The criteria durations and frequencies are biological specifications for protection of aquatic organisms, and their use directly as a design flow using extreme value analysis would result in different frequencies for different receiving waters. Many of these would not be protective. Properly selected design flows, in terms of log Pearson extreme value calculation, may be appropriate for some receiving waters. Other methods for determination of flows are under consideration by the Agency.

Detailed guidance on the calculation of design flows to achieve biologically based water quality criteria will be provided by EPA. Until this guidance is developed, EPA recommends the use of 1Q5 and 7Q5 as CMC and CCC design flows for unstressed systems, and 1Q10 and 7Q10 for stressed systems. A system may be considered to be under stress if there are 1) uncontrolled spills, 2) large areas of impact relative to the size of the waterbody (e.g., because of multiple dischargers), or 3) exposure that will be extended for a longer time period. The longer return frequency should also be used if there is a locally important aquatic species that is known to be particularly sensitive to the type of impact expected (e.g., trout exposed to low dissolved oxygen [DO]). These design flows are considered to be conservative for the vast majority of cases. Design flow regimes other than these have been used under certain circumstances and in qualified site-specific situations. They may continue to be used with proper justification. States should, until publication of the design flow guidance document, contact their EPA Regional Office for the latest information.

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Appendix E.
Lognormal Distribution and Permit Limit Derivations

Introduction

This appendix is intended to provide supporting information for use of the lognormal distribution in technical analysis related to toxics control in general and in permit limit calculations in particular. Descriptive details of this approach are augmented with empirical evidence indicating that the lognormal distribution is the appropriate distribution for permit limit derivations. The concept and calculations are first presented from the standpoint of describing existing effluent quality, and then this framework is applied to derive water quality-based permits. The discussion addresses sampling frequency as it relates to the monthly average daily limit and the treatment of correlated data.

Example Data

To provide some practical insight into the concepts and procedures that follow, we first review three example data sets. These data are actual measured concentrations (in mg/l) of metals in the treated effluent of a foundry and are displayed in Figure E-1. Observe that for each of the metals — lead, copper and zinc — the vast majority of the concentrations tend to be relatively low, but a small, although substantial, number are quite large by comparison. These patterns are typical measurements of pollutants in wastewater, whether they be concentrations, mass loadings, or generic toxicity units (TUs). First, we will examine whether the normal distribution adequately describes these data and, then, whether the lognormal distribution does any better.

Normal Distribution

Most, if not all, of us have encountered the normal probability distribution in a number of applications. Excellent introductions and reviews of this distribution are found in numerous statistical, engineering, and scientific texts, as for example in Reference [1]. Only a terse review will be given here.

Recall that the normal distribution has a probability density function given by

$$f(y) = \frac{1}{\sqrt{2\pi}\sigma} \exp\,[(y-\mu)^2/(2\sigma^2)]$$

for $-\propto < x < \propto$. The symbols μ and σ are the population mean and standard deviation. They, along with other parameters, can be estimated from a set of independent observations $Y_1, Y_2, \ldots, Y_k$ from a normally distributed population according to the formulas in Table E-1.

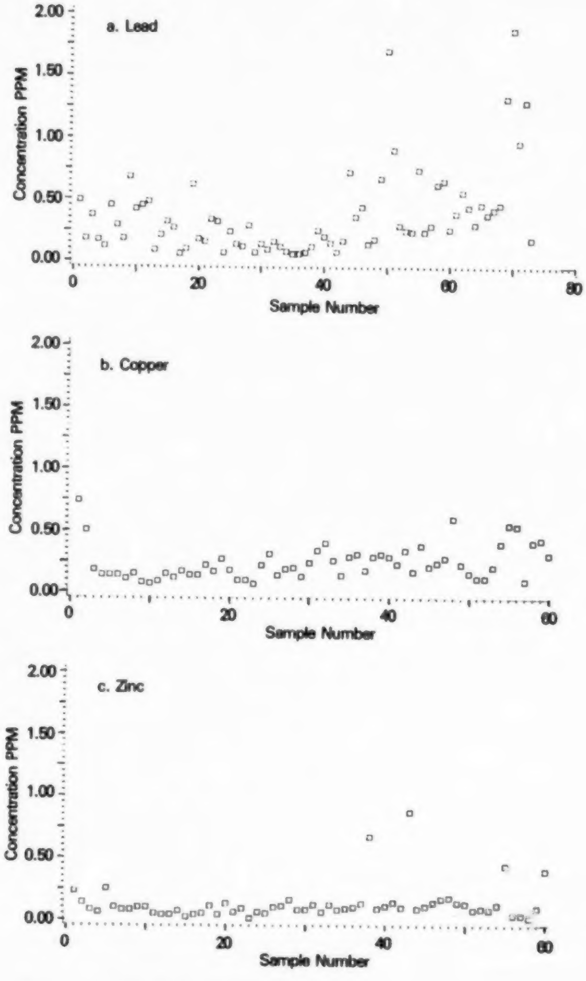

Figure E-1. Effluent concentrations.

The quantity z_p in Table E-1 is the standardized Z-score for the pth percentile given by a table for the normal distribution [1]. That is, z_p is such that the probability of a standardized, normally distributed random variable being less than z_p is p or

$$\text{Prob}\,[Y \leq \mu + \sigma z_p] = p$$

where Y is a normally distributed random variable with mean μ and variance σ^2. The coefficient of variation (CV), the ratio of the standard deviation to the mean, is a dimensionless measure of the relative variability of a distribution. The CV will be used frequently in these procedures, especially in reference to the lognormal distribution.

Table E-1. Population Parameters and Sample Estimates for a Normal Distribution

POPULATION PARAMETER	SAMPLE ESTIMATOR
MEAN = μ	$\overline{Y} = \sum_{i=1}^{k} Y_i/k$
VARIANCE = σ^2	$s^2 = \sum_{i=1}^{k} (Y_i - \overline{Y})^2 / (k-1)$
STANDARD DEVIATION = σ	s
COEFFICIENT OF VARIATION = σ/μ	$s/\overline{Y}$
PERCENTILE = $\mu + z_p\sigma$	$\overline{Y} + z_p s$

Probability and Log-probability Plots

Next, we will review how one can determine whether a normal distribution models a population based on a limited set of measurements. Consider an independent sample of size k, labeled $X_1, X_3, \ldots, X_k$. Let $U_1, \ldots, U_k$ be the sample of Xs in ascending order in which U_1 is the smallest X and U_k is the largest. Now for each U_i, find z_i from the normal table such that

$$P\,[Z \leq z_i] = i/(k+1)$$

and plot each pair (z_i, U_i) on linear graph paper. For example, for the lead data set in Figure E-1, $k=74$. To plot the (z_{50}, u_{50}) set

$$i/(k+1) = 50/(74+1) = 0.66.$$

Now

$$P[Z \leq z_{50}] = 0.66$$

and so $z_{50} = 0.44$, which is the z-value for the 66th percentile from the normal table.

The lead data are plotted in Figure E-2. The straight line in that figure corresponds to the equation

$$u = \overline{x} + zs,$$

where $\overline{x}$ and s are the sample estimates of the mean and standard deviation from the sample. This equation is the best straight line fit to the data based on a normal distributional model. If the data were from a normal distribution, they would fall approximately along the straight line. That is why such plots are called probability plots or normal probability plots. The ordered normal population values are a linear function of the Z-score. Each Z-score is in a one-to-one correspondence to a population percentile. In some probability plots the x-axis is labeled with percentiles instead of Z-scores, although the scale is not linear (one-unit distance is not a fixed increment of the percentiles). In this kind of plot, the order-observed values are plotted with the values $i/(k+1)$, which is the percentile corresponding to the observation.

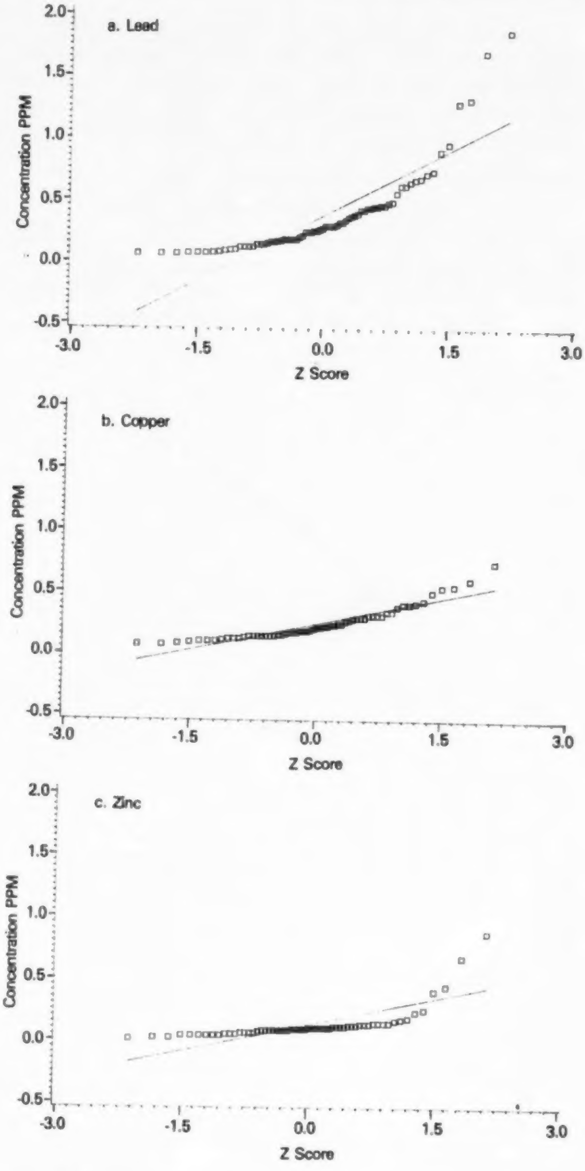

Figure E-2. Normal probability plots.

For the lead data plotted in Figure E-2, the model or straight line underestimates observed measurements for the larger and smaller values; the underestimation of the larger value is the more important in this application to protect water quality. Note that the probability plots for the copper and zinc data exhibit similiar patterns. Do these probability plots indicate that the normal distribution is an adequate model for these effluent data? The answer is given by consideration of an alternative model: the lognormal distribution.

To construct a log-probability plot, set

$$Y_i = \ln(X_i) \text{ or } \log(X_i)$$

for $i = 1, 2, \ldots, k$, and then prepare a probability plot for the Y_i. Here, natural logarithms or logarithms to the base e, denoted by $\ln(x) = \log_e(x)$, are used. Logarithms to the base 10 can also be used.

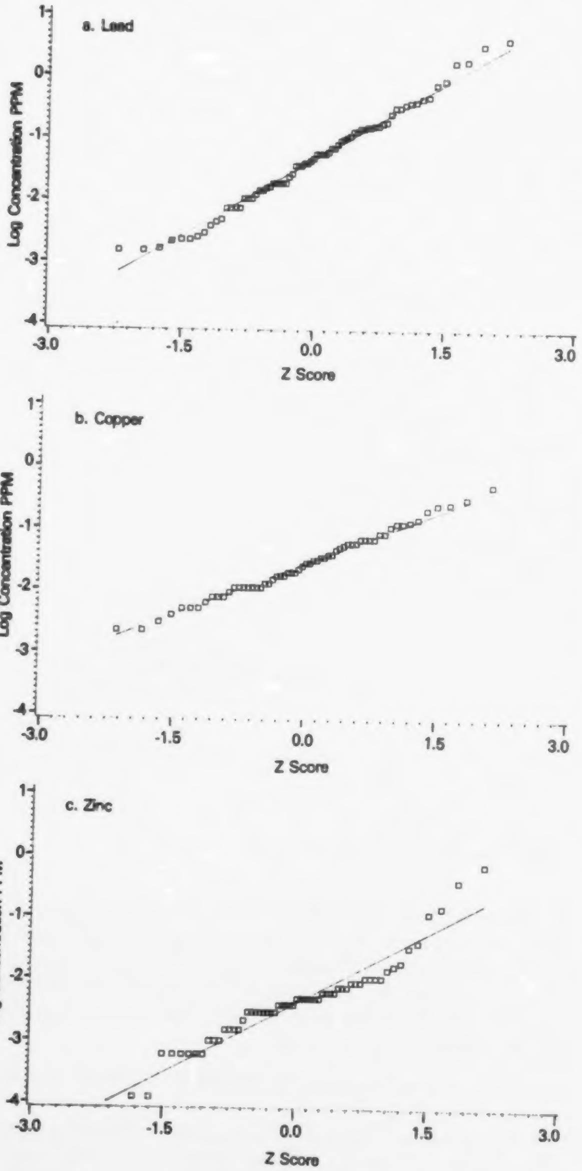

Figure E-3. Lognormal probability plots.

A log-probability plot for the lead data is presented in Figure E-3. Without question, these lead data follow a straight line on the log-probability plot much more closely than the same data plotted on the probability plot in Figure E-2. The straight line in the log-probability plot is the best-fit model based on a lognormal distribution. The log-probability plots for the copper and zinc data sets exhibit the same pattern.

This empirical analysis demonstrates that these data sets follow a lognormal distribution. This finding is not peculiar to these data in particular. Experience has shown that the same patterns emerge in analyses of effluent concentrations and mass loadings. The degree of fit for the lognormal distribution does vary somewhat from application to application, but not enough to alter the conclusion that effluent pollutant discharges are generally lognormally distributed.

Lognormal Distribution

A random variable X behaves according to the lognormal distribution if the random variable $Y = \ln(X)$ is normally distributed. Effluent data are lognormally distributed. Therefore, the analysis procedures for analyzing lognormal data are similar to those for the normal distribution once the logarithmic transformation has been applied. The analysis procedure is:

- Transform the observation to the log-space.
- Perform analyses and obtain results.
- Transform the results for the log-space back to the observation-space.

The logarithmic transformation introduces specific relationships between the parameters in the normal and lognormal distributions. For instance, when X is lognormally distributed, then $Y = \ln(X)$ is normally distributed. If Y has mean μ and variance σ^2, then the mean of X is given by

$$E(X) = e^{\mu + \sigma^2/2}.$$

Other relationships between the lognormal distribution of X in the observation-space and the normal distribution of $Y = \ln(X)$ in the log-space are given in Table E-2.

Table E-2. Normal and Lognormal Distributions

	Y-NORMAL (log-space)	X-LOGNORMAL (observation space)
VARIABLE	$Y = \ln(X)$	$X = \exp(Y) = e^Y$
POPULATION PARAMETER		
MEAN	μ	$e^{\mu + \sigma^2/2}$
MEDIAN	μ	e^{μ}
VARIANCE	σ^2	$e^{2\mu + \sigma^2}(e^{\sigma^2} - 1)$
STANDARD DEVIATION	σ	$e^{\mu + \sigma^2/2}(e^{\sigma^2} - 1)^{1/2}$
COEFFICIENT OF VARIATION	σ/μ	$(e^{\sigma^2} - 1)^{1/2}$

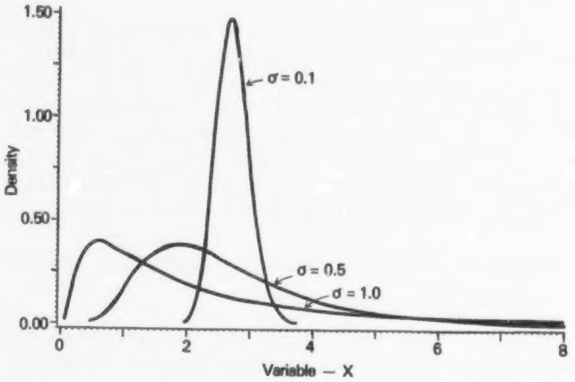

Figure E-4. Examples of lognormal densities.

The shape of the lognormal probability density function is determined by the parameter σ, which is the standard deviation in the log-space. For a relatively small value of σ, say $\sigma = 0.1$, the lognormal density appears, at least to the eye, symmetric. For relatively large values of $\sigma \geq 1$, it is highly skewed to the right. In the kinds of applications of interest here, the distributions are almost always highly skewed, and the values of σ are generally in the range of $0.2 < \sigma < 1.2$, although there are exceptions. Illustrations of lognormal probability density functions are provided in Figure E-4.

Examples of the use of the lognormal distribution are given in subsequent sections. In these sections, the data are first transformed to the log-space, and then the parameters μ and σ are estimated from the transformed data. An alternative approach is to estimate the mean and variance in the observation-space and then derive estimates of μ and σ by use of the formulas in Table E-2. The approach based on transformed data used in this section is preferred in that the sampling errors of the population estimates are minimized from the theoretical standpoint. The alternative, however, is often employed in practice.

Daily Permit Limit

The daily maximum permit limit should be the 95th upper percentile value of the pollutant distribution that will comply with legal requirements. In certain cases the 99th percentile value may be allowable. For the daily maximum limit we assume the observed discharge concentrations (or mass loadings or generic TUs) are independent samples from a lognormal distribution.

Consider the lead concentration data (in mg/l) presented in the Example Data section, Figure E-1. There are 74 values ($k = 74$); denote the observed concentrations by x_i and set $y_i = \ln(x_i)$. Using the estimation formulas in Table E-1, we have

$$\bar{y} = \sum_{i=1}^{74} y_i/74$$

$$= -1.30$$

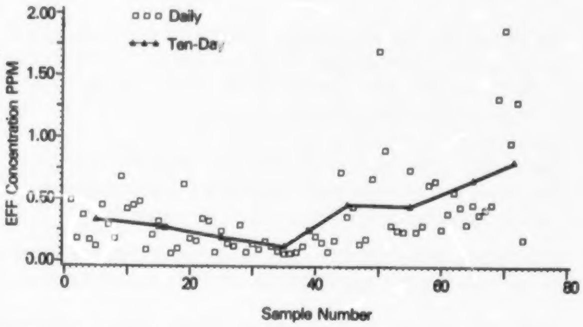

Figure E-5. Daily and 10-day average: Lead data.

and

$$s_y^2 = \sum_{i=1}^{74} (y_i - \bar{y})^2/73$$

$$= (1.03)^2.$$

By transforming these parameter estimates back to the observation-space, the estimates of the mean and standard deviation of the concentration distribution are found to be 0.46 mg/l and 0.63 mg/l, respectively. The estimated coefficient of variation, therefore, is 1.37.

From Table E-1, the estimate of an upper percentile of the lognormal distribution is given by

$$\exp(\bar{y} + z_p\, s_y) \text{ mg/l}.$$

Therefore,

$$\text{95th limit } (z = 1.645) = 1.48 \text{ mg/l}$$

and

$$\text{99th limit } (z = 2.326) = 2.99 \text{ mg/l}.$$

Monthly Permit Limit

The monthly permit limit is the 95th or 99th percentile of the distribution of the arithmetic mean of the daily effluent values. The distribution is based on the assumed number of samples that will be taken during each month. This assumed number is generally recommended to be 10 samples per month.

We again consider the lead concentration data that were used for the daily maximum limit example in the previous section. To illustrate the effects of averaging observations, the data are displayed again in Figure E-5 together with averages of successive subsets of size 10 (i.e., in that figure, samples 1 through 10 were averaged; samples 11 through 20 were averaged, and so on). The average values are connected by a straight line. Observe that these averages exhibit much less variation than the individual data. To obtain the monthly limit, we calculate the percentile value of these averages.

Again, we assume the daily observations are independent and are drawn from a lognormal distribution. If we assumed that n samples are taken each month, then the percentile for the distribution of monthly averages is computed according to the table below. The formulas in the table are based on the assumption that the sum of lognormally distributed random variables is distributed according to a lognormal distribution. Theoretically this assumption is not true, but the formulas, nonetheless, provide a very accurate approximation as demonstrated in Reference [3] and elsewhere.

The formulas for computing monthly averages based on an assumed n samples per month are as follows:

Table E-3. Formulas for Computing Monthly Averages Based on An Assumed k Samples Per Month.

pth percentile $= \exp(\bar{y}_n + z_p s_n)$

where $s^2_n = \ln(1 + (e^{s^2} - 1)/n)$

$$\bar{y}_n = \bar{y} + (s^2 - s^2_n)/2$$

and $\bar{y}$ and s are as in previous section

To illustrate the monthly limit calculations, consider the lead data set used in the previous section to calculate the daily permit limit. Recall for those data

$$k = 74.$$

$$\bar{y} \ -1.30.$$

$$s = 1.03.$$

Using n=10, from the equations above we have

$$s_{10} = 0.42 \text{ and}$$

$$\bar{y}_{10} = 0.86.$$

The 95th percentile limit ($z_{95} = 1.645$) is therefore

$$\exp(\bar{y}_{10} + (0.42)(1.645)) = 0.85 \text{ mg/l},$$

which is considerably less than the daily 95th percentile limit for the same data set of 1.48 mg/l.

Comment on Independence Assumption

In this and in the previous section, we have assumed that all the observed pollutant levels are independent or uncorrelated. In the case of the monthly limit derivation, this assumption can be quite critical. If the effluent levels are correlated, the actual monthly 95th percentile can be substantially higher than that derived from the analysis based on the independence assumption. There is essentially no effect, however, on the calculated daily limit validity. Whenever the data show a positive correlation, that correlation should be factored into the limitation; correlated or dependent data are addressed in the subsection entitled Correlated Data.

Effect of Assumed Number of Samples Per Month

The monthly limit procedures presented in this section allow one to derive a monthly limit for an assumed number of samples taken during the month, denoted by n. One can calculate a 95th percentile for n=1, for n=2, ..., and for n=30. How do the results behave as a function of n? For each of three pollutants, monthly limitations for each value of an assumed number of samples are presented in Figure E-6. The general pattern is that the calculated limitations are relatively insensitive to n for n ≥ 8. This analysis was used in part to support the general recommendation that n should not be less than 10 for pollutants that are inexpensive to monitor.

Effect of Actual Number of Samples Taken During the Month

Whatever value of n is selected to determine the monthly limitation, the discharger may not take exactly n samples during every month. For example, we derived 95th percentiles based on the assumption of 10 samples per month. What is the effect of the permittee taking a different number of samples, say 12, during the month? This effect can be assessed by calculating the theoretical probability that the monthly average (95th percentile in this example) based on n=10. This probability can be calculated for any number of samples taken. These calculations are displayed for this example in Figure E-7; the actual number of samples is plotted on the X-axis, and the percentile is plotted on the Y-axis. Observe that the calculated probability is relatively insensitive unless the actual number of samples is substantially less than 10.

Consistency of Limits

It is important to emphasize that these procedures provide consistent daily and monthly limits. They are consistent in that they both incorporate the same level of effluent quality and they are equally stringent. Enforceability is enhanced because both limitations are equally stringent. Limitations that dictate different levels of performance are avoided. Accordingly, the permittee can be expected to meet both the daily and monthly limitations together.

Derivation of Water Quality-based Limits

Technology-based effluent quality limits (BAT, Best Conventional Technology [BCT], secondary) are derived from treatment system performance. Water quality-based limits are derived from the required treatment system performance necessary to comply with the wasteload allocation (WLA). The mathematical expressions for water quality limits are the same as those for technology-based effluent quality limits; the major difference is that the parameters in those expressions are derived from the WLA. The procedure is illustrated by an example.

We assume that for the existing treatment system, which may or may not satisfy the WLA requirements, the level of performance is known. Returing to the lead data example, we have from the section on Daily Permit Limits:

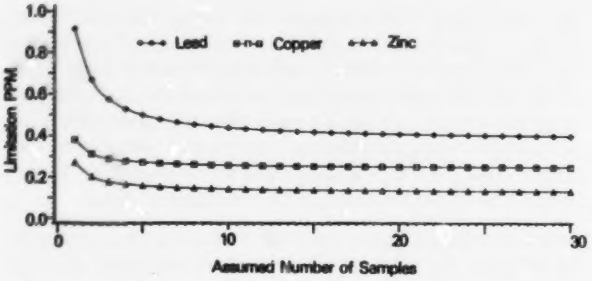

Figure E-6. Limitations as a function of assumed number of samples.

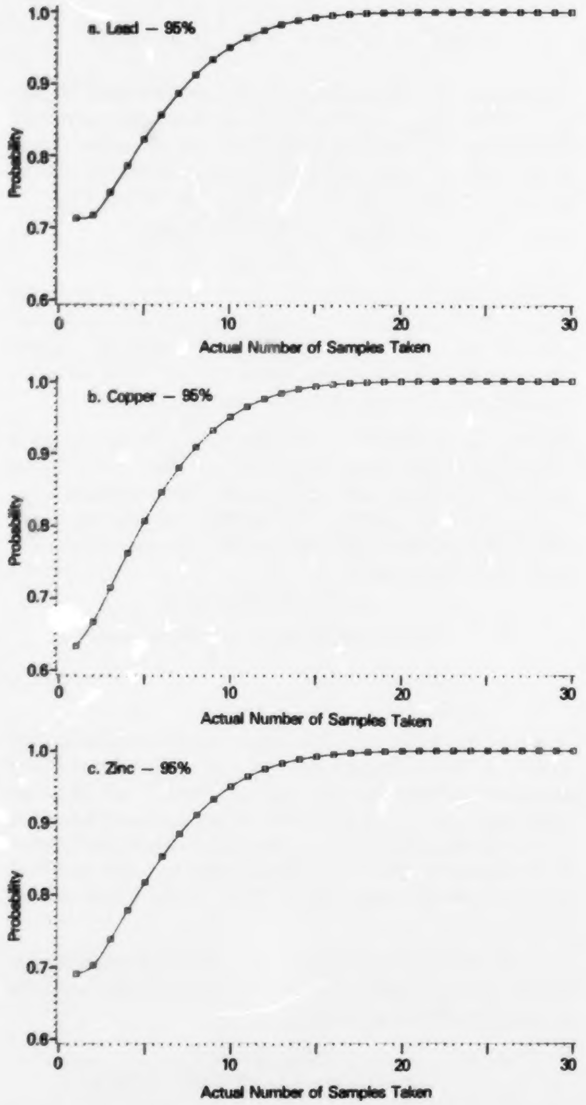

Figure E-7. Probability associated with monthly averages.

$$\text{LTA} = 0.46 \text{ mg/l}$$

and

$$\text{CV} = 1.37,$$

where LTA is the long term average. We assume further, for this example, that new requirements can be satisfied by a fixed percentage reduction in effluent loadings, say by addition of a filter. This kind of modification will have the impact of reducing the LTA but will not change the CV. (Other kinds of modifications, such as the addition of a polishing pond would reduce the CV, which should be incorporated into the analysis.)

Suppose the acute WLA requirment is a one-day (composite or grab sample) concentration requirement given as

$$\text{Acute WLA} = 0.50 \text{ mg/l of lead}$$

and the requirement is to be satisfied 99% of the time. We must, therefore, satisfy

$$\exp(\mu_a + z_{99}\,\sigma) = 0.50 \text{ mg/l}.$$

Now $z_{99} = 2.326$ and

$$\sigma^2 = \ln((\text{CV})^2 + 1)$$
$$= (1.03)^2$$

is known and fixed. We must find a μ that satisfies the above equation. We find that

$$\mu_a = -3.08,$$

which leads to a required LTA of

$$\text{LTA}_a = \exp(\mu_a + \sigma^2/2)$$
$$= 0.077 \text{ mg/l}$$

which is a reduction in effluent lead loading of

$$(1-0.077/0.46) \times 100 = 83\%$$

for the acute WLA requirement. The corresponding 95th percentile limits are

Daily 95% limit = 0.25 mg/l
Monthly 95% limit = 0.093 mg/l,

which are derived from μ_a and σ and assuming 10 samples are taken per month.

In addition to the acute WLA, the chronic WLA requirement of

$$\text{Chronic WLA} = 0.05 \text{ mg/l} - \text{seven-day average},$$

which must be satisfied 99% of the time. Because this chronic WLA must be satisfied on average for a seven-day period we set

$$\exp(\mu_7 + z_{99}\,\sigma_7) = 0.05 \text{ mg/l},$$

which will ensure that the average for each day over a seven-day period will be met 99% of the time. Now σ is known, and we can calculate μ_7 and then μ_c. The required LTA for the chronic WLA is

$$\text{LTA}_c = \exp(\mu_c + \sigma^2/2)$$
$$= 0.032 \text{ mg/l},$$

a 93% reduction in the existing level of lead discharged. Because reduction induced by the chronic WLA is more stringent than that of the acute (93% vs. 83%), the daily and monthly limits are based on

$$\mathrm{LTA}_c = 0.032 \text{ mg/l}$$

and

$$\mathrm{CV} = 1.37.$$

The corresponding 95th percentile limits are

$$\text{Daily limit} = 0.104 \text{ mg/l}$$

and

$$\text{Monthly limit} = 0.051 \text{ mg/l},$$

assuming 10 samples per month.

These limits are internally consistent and ensure that both the acute and chronic WLA will be satisfied.

Correlated Data

The analysis in this Appendix to this point has been predicated on the assumption that measured effluent pollutant loadings are independent or uncorrelated. If data are strongly correlated, which is frequently the case with POTWs, the monthly permits can be in substantial error if the correlation is not taken into account in the analysis. In this section we address the incorporation of dependence in the calculations.

A major factor that determines whether effluent levels are highly correlated is the retention time of the wastewater treatment system. If the retention time is large relative to the time between effluent samples, then those samples will be correlated in most cases. In municipal systems, for example, the retention time is frequently a matter of days, and sampling is often conducted on a daily basis. The effluent levels, consequently, are substantially correlated. However, in many industrial systems, for instance a physical/chemical treatment system for electroplating wastewaters, the treatment system retention time is relatively short: four to eight hours. Daily effluent levels from these kinds of systems are generally uncorrelated or statistically independent. These general patterns are the same irrespective of the kind of pollutant in question.

To illustrate the importance of accounting for correlation in these analyses, we consider an example. In Section C of Reference [4], an aggregate coefficient of variation for public treatment works with activated sludge is given as 0.85 for daily TSS concentrations. The average TSS for the same plants is 14.3 mg/l. The standard deviation for daily TSS is, consequently,

$$(14.3)(0.85) = 12.15 \text{ mg/l}.$$

If the daily TSS concentrations are independent, then the standard deviation of a 30-day average (30 samples) would be

$$12.15 \div \sqrt{30} = 2.22 \text{ mg/l}.$$

However, for these same plants, the coefficient of variation for a 30-day average is reported to be 0.45, yielding a standard deviation of

$$(14.3)(0.45) = 6.43 \text{ mg/l}$$

for the 30-day TSS average. This value is almost three times larger than that value based on the assumption of independence, which indicates that the daily TSS levels are strongly correlated. Because the actual standard deviation, which incorporates the correlation, is so much larger than that based on the independence assumption, it must be reflected in the analysis. Otherwise, the results would be in substantial error.

The theoretical framework for including correlation is as follows. Recall that if the random variables $X_1, X_2, \ldots, X_n$ are independent with variance σ_x^2 then

$$\mathrm{Var}(\overline{X}) = \sigma_x^2/n$$

where $\overline{X}$ is the average of $X_1, X_2, \ldots, X_n$. If the X_i are correlated then

$$\mathrm{Var}(\overline{X}) = \sigma_x^2[1/n + 2/n^2 \sum_{j=1}^{n-1} (n-j)\varrho_j]$$

where ϱ_j is the correlation coefficient between X_i and X_{i+j}. When the ϱ_j are positive they have the effect of increasing the variance of $\overline{X}$. The ϱ_j, in principle, range from -1 to $+1$, but in this application the coefficients are almost always positive. The ϱ_j are defined by

$$\varrho_j = \frac{E[X_{i+j} - E(X_i)]\,[X_i - E(X_i)]}{\mathrm{Var}(X_i)}$$

If the correlation coefficient $\varrho_j = 1$, then $X_i = X_{i+j}$, but if $\varrho_j = 0$, then X_i and X_{i+j} are independent or uncorrelated. As ϱ_j increases from 0 to 1, then X_{i+j} tends towards X_i. In the extreme case of $\varrho_j = 1$, there is no reduction in variance due to averaging.

Having an estimate of $\mathrm{Var}(\overline{X})$, one can compute a monthly average limit using the formulas to compute the monthly limit given previously. For example, consider a monthly average limit based on an assumed 30 samples per month. We first would compute from the numerical illustration

$$n_e = \frac{\text{daily variance of } X_i}{\text{actual variance of 30-day average}}$$

$$= (12.15)^2/(6.43)^2$$

$$= 3.57$$

and use the value n_e in the lognormal formulas rather than $n = 30$. Although there are actually 30 samples included in this 30-day average, there are only an equivalent of $n_e = 3.57$ independent samples because of the strong correlation. Using the value n_e instead of n incorporates the correlation into the calculations involving an average. Note, if the X_i are uncorrelated then $n_e = n$.

This general approach allows one to incorporate dependence into distributional calculations for an n-sample average by first computing

$$n_e = \frac{\text{daily variance}}{\text{actual n-sample average variance}}$$

where these variances are of the actual pollutant levels in the observation-space, not the log-space.

Estimating n_e from Correlated Data

To compute the actual n-sample variance when daily plant data are available, say $X_1, X_2, \ldots, X_k$, one can compute the cross-correlations and utilize the formula for the variance given previously in this subsection. If for instance $n = 10$, and if we assume a sample is taken every third day, the cross-correlations $\varrho_3, \varrho_6, \ldots, \varrho_{30}$ would be required.

Alternatively, one can very much reduce the computation and data requirements by introducing a simple model that states that the effluent concentration on day i is linearly related to the previous day's concentration plus a random variation. This model is expressed as

$$X_i = \varrho X_{i-1} + \epsilon_{i-1},$$

where ϵ is an independent error term. This model (first-order autoregressive model — see Reference [6]) has been applied to estimate effective average sample sizes for effluent data [5,7]. Using this model one can estimate n_e directly by

$$n_e = \frac{n^2(1-\varrho)^2}{n(1-\varrho^2) - 2\varrho(1-\varrho^n)},$$

where ϱ, the lag-1 correlation, is estimated by

$$\frac{\sum_{i=1}^{k-1} (x_i - \overline{x})(x_{i+1} - \overline{x})}{\sum_{i=1}^{k} (x_i - \overline{x})^2}$$

This formulation is for the n samples taken on successive days during the month. It, however, can be modified. For example, assume that n samples are taken during the month and a sample is taken every third day. Then the adjusted formula is

$$n_e = \frac{(10)^2\,(1-\varrho^3)^2}{10(1-\varrho^6) - 2\varrho^3(1-\varrho^{30})},$$

which is obtained by substituting ϱ^3 for ϱ because for the given model the correlation between X_i and X_{i+j} is ϱ^j. Suppose, for instance, $\varrho = 0.80$ is an estimate of the lag-1 correlation derived from daily data. Then

$$n_e = \frac{(100)(0.238)}{7.378 - 1.022}$$

$$= 3.74$$

The 10-sample average has an effective sample size of 3.74.

In the subsection entitled Monthly Permit Limit we calculated a monthly 95th percentile lead limit of 0.85 mg/l based on an assumed 10 samples a month. That calculation was based on the data being independent. If, however, the data are correlated and the effective sample size of $n_e = 3.74$, we have

$$s_{3.74} = 0.639,$$

and

$$\overline{y}_{3.74} = -0.974,$$

and the 95th percentile monthly limit becomes

$$\exp(-0.974 + (1.645)(0.639)) = 1.08 \text{ mg/l}.$$

The effect of this correlation is to increase the monthly limit from 0.85 mg/l to 1.08 mg/l. If daily monitoring were assumed (i.e., $n = 30$), the relative increase due to the correlation would have been substantially larger. Conversely, if a low sampling frequency were assumed, say $n = 4$, the difference between assuming independence and incorporating correlation would have been slight.

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